The Recovery of Benthic Communities Following Organic Enrichment: Examples from Caged Finfish Aquaculture.

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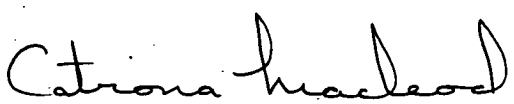
Declarations

Statement of originality

Except as stated herein this thesis contains no material which has been accepted for the award of any higher degree or diploma by the University or any other institution. To the best of my knowledge and belief this thesis contains no material previously published or written by another person except where due acknowledgement is made in the text.



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Abstract

A clear understanding of the factors affecting recovery processes is required for effective management of the impacts of organic enrichment on marine sedimentary environments. Using finfish aquaculture as an example this study investigated the recovery response in southern temperate sediments, identifying critical processes and indicators that could be used to improve management practices. Ecological and geochemical responses and the effect of regional differences in the extent and rate of recovery were also examined over a range of timescales of recovery (i.e. short and long-term).

In general the pattern of recovery followed established successional principles. Initial recovery was rapid following cessation of organic input. After only 2 months a marked improvement in sediment condition could be discerned. Benthic communities responded more slowly than sediment geochemistry. After 36 months the benthic infauna below the cages still differed from the references even though other sediment measures had recovered. Nonetheless, the long-term study indicated that the system had functionally recovered after only 12 months and benthic communities contained a diverse range of species with broad ecological sensitivities. Once the ecological function of the sediment was restored subsequent changes in the community structure were relatively minor, reflecting the addition of rarer climax species with longer reproductive cycles and more sensitive larval stages. Comparison of different approaches for evaluation of recovery revealed that the physico-chemical measures routinely employed in measurement of impact are of limited use in assessment of recovery.

In a comparison of recovery response over 3 months at two different study locations it was found that rate and extent of recovery were affected by location, initial impact of the sediments, and length of fallow period. Initial recovery was faster at the more sheltered site than at the more exposed site, possibly reflecting differences in environmental resilience with the more sheltered location better able to assimilate organic inputs. Sediments at the more sheltered site had naturally high organic carbon content and there was greater similarity in ecological function between unimpacted and

impacted conditions. In contrast, at the more exposed site the sediments had a very low organic content and ecological function was significantly altered after impact. The natural fauna at this site was less able to re-establish directly by immigration, and relied to a greater extent on interim remediation of the sediments by transitional species. This has important implications for environmental management, as it suggests that the sediments in some areas have a greater natural resilience to organic inputs.

Together the results of the present study have increased our understanding of the recovery processes associated with organic enrichment in southern temperate regions and indicate that, since recovery response differs depending on the background environmental conditions, establishment of baseline conditions and local benchmarks is essential in evaluation of impact and recovery, for establishment of a regulatory framework and for ongoing environmental management. However, these baselines and the subsequent management protocols must be established at a spatial scale relevant to the community (ecological) changes.

Statement of co-authorship

Chapters 2-5 of this thesis have been prepared as scientific manuscripts. In all cases experimental design and implementation of the research program, data analysis, interpretation of results and manuscript preparation were the primary responsibility of the candidate, but were carried out in consultation with supervisors, and with the assistance of co-workers. Contributions of co-authors are outlined below:

Chapter 2 (Paper 1): Assessment of long term change in sediment condition after organic enrichment: defining recovery. *Catriona Macleod (70%), Dr Christine Crawford (15%), Dr Natalie Moltschaniwskyj (15%)*

Chapter 3 (Paper 2): Ecological and Functional Changes Associated with Long-Term Recovery from Organic Enrichment. *Catriona Macleod (80%), Dr Natalie Moltschaniwskyj (15%), Dr Christine Crawford (5%)*

Chapter 4 (Paper 3): Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. *Catriona Macleod (75%), Dr Natalie Moltschaniwskyj (20%), Dr Christine Crawford (5%)*

Chapter 5 (Paper 4): Biological Recovery from Organic Enrichment associated with Finfish Cage Aquaculture: Do Some Systems Cope Better than Others? *Catriona Macleod (75%), Dr Natalie Moltschaniwskyj (15%), Dr Christine Crawford (5%), Susan Forbes (5%)*

Dr Natalie Moltschaniwskyj (Tasmanian Aquaculture and Fisheries Institute) was the primary supervisor for this Ph.D. study and as such has contributed to the development of ideas as well as providing analytical advice and assistance. As a research supervisor Dr Christine Crawford (Tasmanian Aquaculture and Fisheries Institute) also contributed to the research ideas, their formalization and development. In

addition both Dr Moltschaniwskyj and Dr Crawford assisted with the refinement and presentation of the publications. Susan Forbes provided general laboratory assistance on paper 4.

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Finally I need to say a huge thanks to my family, Harry, Kenny and Jamie. My two boys, Kenny and Jamie, made a huge contribution to the completion of this thesis by "not bugging Mummy when she's working". Harry's support, patience and encouragement got me through the most difficult stages and his proof reading abilities are second to none. You are "on a promise" - we're ALL going to the beach next summer!!

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Chapter 1:

General Introduction

1.1 Introduction

There is a considerable body of research examining the effects of organic enrichment on benthic communities but by far the greatest component of this has focussed on the impacts with considerably less research on the recovery response, particularly in boreal areas (Black, 2001). Even fewer studies have attempted to evaluate the rate at which the sediments recover or the effect of environmental conditions on the recovery process. Consequently the overall objective of this study was to characterize the key factors influencing benthic community response in sediments recovering from organic enrichment.

The macrobenthic successional stages associated with an increasing organic enrichment gradient were defined in the "classical" study of Pearson and Rosenberg (1978) (Fig. 1.1). This model has been validated in many subsequent investigations for a variety of organic enrichment sources including aquaculture (eg. Brown *et al.*, 1987; Ritz *et al.*, 1989; Weston, 1990; Holmer and Kristensen, 1992; Findlay *et al.*,1995; Cheshire et al., 1996; Hargrave, *et al.*, 1997, Karakassis *et al.*, 1999, Wildish *et al.*, 2001, Macleod *et al.* 2002, Brooks *et al.*, 2003). Although the successional changes involved in the degradation and recovery processes have been shown to be similar (Black, 2001), the rates at which they occur are likely to differ, since sediment recovery is a passive process whilst degradation involves the active input of organic material. Criteria associated with evaluating impact may not be directly transferable to the assessment of recovery and specific recovery benchmarks have not yet been established.

Although characterisation of benthic infaunal communities is one of the most reliable indicators of sediment condition, it is expensive and time consuming (Wildish *et al.*, 1999; Crawford *et al.*, 2002). Several studies have compared the infaunal successional categories defined by Pearson and Rosenberg (1978) to other physical-chemical and biological parameters (Brown *et al.*, 1987, Weston, 1990, Holmer and Kristensen, 1992, Findlay *et al.*, 1995, Cheshire *et al.*, 1996, Hargrave *et al.* 1997, Wildish *et al.*, 2001, Macleod *et al.* 2002) and have suggested a direct relationship between the chemical status of the sediment and the infaunal community structure. As a

result these simpler and more cost effective chemical techniques are frequently used as surrogate measures of sediment condition (Hargrave *et al.*, 1997; Kingsford and Battershill, 1998). Some of these measures (e.g. redox and sulphide) indicate very specific chemical aspects of sediment condition (Holmer and Kristensen, 1992; Hargrave *et al.*, 1993). In addition visual assessment of sediment characteristics and epibiota by diver or video are also relatively common these days (Krost *et al.*, 1994, Angel *et al.*, 1998, Crawford *et al.*, 2002). However, the value of many of these approaches in relation to sediment recovery is yet to be clarified.

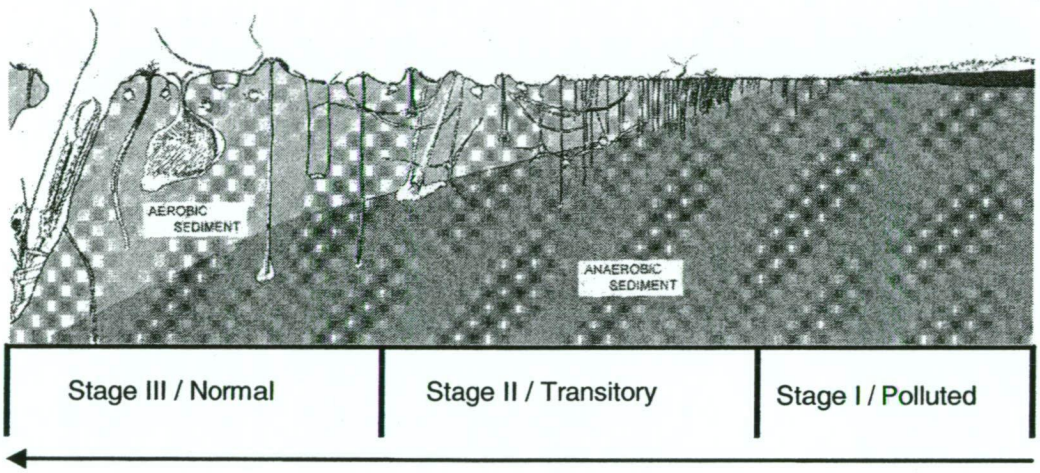


Fig. 1.1 Adaptation of Pearson and Rosenberg (1978) and Rhoads *et al.* (1978) models (modified from Black, 2001).

1.1.1 Defining recovery

Defining what constitutes recovery is critical to any assessment of the rate or extent of recovery. There are many different ways to define recovery; it may be a return to "pristine" conditions or to some other pre-determined level, it could be restoration of sediment chemistry or biology, re-establishment of species diversity, species number or faunal abundance, restoration of particular species or community types or full community equivalence with unimpacted conditions. All of these definitions are perfectly valid, however, each approach will give a different estimation of the environmental condition. Accordingly it is critical that the limitations of any given approach are fully understood. Geochemical recovery of the sediments is

generally more rapid than biological recovery (e.g. Carroll *et al.*, 2003). Similarly, using only selected components of the community to determine recovery may also result in a low evaluation of recovery time whereas full community assessment may result in a more conservative estimate. To make management decisions regarding recovery it is important to be clear on the level of recovery upon which those decisions are being made. Consequently, it is very important when comparing recovery that the underlying measure is equivalent and to establish "a priori" what precisely is meant by recovery.

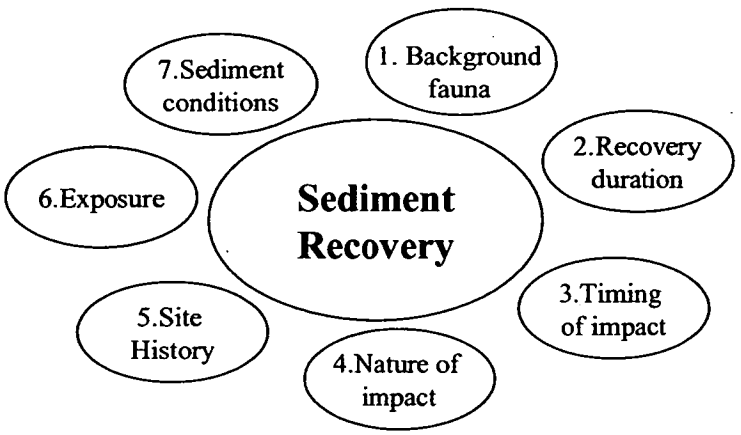
Recovery is a complex process and the factors affecting this are highly interrelated (Fig. 1.2). Clearly the nature, frequency and size of the impact will have important consequences for subsequent recovery (Zajac *et al.*, 1998). Not all components of an ecosystem will be affected equally by a given impact, some may be more affected considerably more than others, and this may have an important affect on the systems ability to recover. Such conditions would also have an important effect on the appropriateness of any measure of recovery. Some disturbances may be so severe that the environment is completely and permanently changed, an example of this might be an area subject to landfill. The temporal nature of the impact will also have an important effect on the recovery rate. Chronic and sustained impacts on ecosystem structure or function (press disturbances), such as from a continuous pollution source, have been shown to have quite a different recovery response to that of short-term pulse disturbances, such as contaminant spills or floods (Power, 1999). The resilience of the system, the rate at which measured responses return to pre-disturbance levels (Power, 1999), will vary with the type and magnitude of disturbance (Gore *et al.*, 1990). The biological attributes and ecology of the surrounding area play a significant role in the recovery response (Niemi et al., 1990).

The recovery process is also strongly influenced by the prevailing environmental conditions; site characteristics such as water depth, particle size, current velocities, and tidal effects play an important role in determining the rate and extent of recovery. Hydrodynamically energetic areas tend to be erosional with relatively low residence times for waste materials, whereas sheltered quiescent locations tend to be depositional in nature (Black, 2001). This will influence the grain size and particle

distribution (Hall, 1994) which will in turn significantly affect the benthic ecology, determining the community composition and ultimately affecting the system's capacity to respond to environmental stressors (Snelgrove and Butman, 1994).

Ecosystem theory suggests that the ecosystem is a complex and stable self-regulating system which has evolved mechanisms for self-repair (O'Neil, 1999). Populations within an ecosystem are adapted to resist and recover from random fluctuations in the environment (Odum 1969). However, there is also a view that if subjected to sufficient disturbance ecosystems can jump to a new configuration where, although recovery may still occur, it does not return to the original ecosystem (Holling, 1973, Young *et al.*, 2001). This clearly has significant implications for assessment of recovery.

Critical to the recovery process is the ability of species to colonise the recovering area. Where the disturbance is highly localised then direct migration from surrounding areas may occur. Species which rely on broadcast spawning may recover rapidly after localised short-term disturbances (Skilleter, 1995). However, the effectiveness of this approach will be limited by the availability of reproductive adults in the region and the extent of larval dispersal. Species interactions may facilitate or inhibit successional changes (Connell and Slatyer, 1977). These interactions may differ depending on the timing of the disturbance event; temperature and season will affect the biology and ecology of the system, particularly influencing reproductive state and resource availability. The presence of introduced species can radically change normal species interactions and alter successional colonization patterns. To effectively manage recovery it is essential to have an understanding of the influence of all these factors on the recovery process



	Main Influences on Sediment Recovery
1.	*Recruitment/recolonisation process, bioturbation*
2.	*Short or long term will affect recovery level attained*
3.	*Time at which impact occurs may affect sediment chemistry and faunal ecology (i.e. life history stages)*
4.	*Single or continuous impact; combined stressors (eg. enrichment and physical disturbance)*
5.	*Previous impacts and other uses*
6.	*Hydrodynamic environment will affect sediment conditions and faunal composition*
7.	*Faunal composition, sediment geochemistry*

Fig. 1.2 Main factors influencing recovery process.

1.1.2 Anthropogenic impacts & environmental sustainability

Public concern regarding the effects of development on the marine environment has increased significantly in recent times. As a consequence environmental sustainability and ecosystem health are now critical considerations for environmental management. Determining the extent to which any impact is sustainable requires an understanding of the extent to which the system is able to recover after the impact is removed. Organic enrichment is one of the most common anthropogenic impacts in the marine environment; sewage treatment outfalls, wood pulp mill effluents and caged fin-fish aquaculture are all relatively common sources of localized organic enrichment.

1.1.3 Marine fin-fish aquaculture

A global increase in aquaculture production is predicted over the next 15 years

with estimates suggesting that total production will double from current production levels (FAO, 2004). Aquaculture is the fastest growing primary industry in Australia. In 2003/4 total aquaculture production in Australia was valued at $732 million and is expected to increase to ca. $2.5 billion by 2010 (Dept. Agriculture, Fisheries & Forestry, 2006). In Tasmania Atlantic salmon (*Salmo salar* L.) culture was valued in excess of $110 million in 2001/2 (Love and Langenkamp, 2003) and provides significant employment opportunities in regional areas. The Tasmanian government recognises the economic and social benefits associated with a productive aquaculture industry and is highly supportive of its further development, provided it is ecologically sustainable. The salmon aquaculture industry recognises that to be economically sustainable it needs to be environmentally sustainable, and that to do this it needs to have a clearer understanding of the relationship between farming practices and environmental conditions. Information on the effectiveness of fallowing as a means of rehabilitating sediments is vital for the optimal management of lease areas and to ensure that production is sustainable. Consequently, it was hoped that this project would assess the rate of recovery associated with fallowing practices and determine if current farming practices were sustainable.

It is well recognised that one of the most significant impacts from caged fish farming is the organic enrichment of the sediments as a result of faecal waste and uneaten feed (Iwama, 1991, Black *et al.*, 2001). What is less well understood is how sediment conditions are affected by farming practices, eg rotational farming/fallowing, and different background environmental conditions. To alleviate impacts on the sediments and to give the sediment an opportunity to recover, fish-holding cages are often removed or are left unstocked for a period of time. Carroll *et al.* (2003) suggest that recovery of sites by periodic abandonment (fallowing) is one of the best management tools for sustainable salmon farming in cold-water environments. Fallowing practices in Tasmania tend to be different to those employed by salmon farming operations in other countries, because the production stocking densities are relatively low (typically < 15 kg m^{-3}; Tasmanian Salmon Growers Association, pers. comm.) compared to operations elsewhere (up to 25 kg m^{-3}; Willoughby, 1999). In addition the scale of the industry is relatively small so that although some farms may

remove all cages from a lease, more often only sections of the lease or certain cage sites are fallowed, while the remainder of the farm remains operational. In contrast, salmon operations in other countries commonly fallow whole leases/systems (Beveridge, 1987). Although environmental monitoring of the sediments is mandatory, there is no legislative requirement for fallowing in Tasmania. Environmental regulations only require that there be no "unacceptable impact" associated with farming practices (Woods *et al.*, 2004). Therefore the duration of any fallow period is largely at the discretion of the farm manager. For the most part the fallowing time is determined by economic/production pressures, company policy, and the farmer's experience. However, the amount of time actually required for sediment recovery is poorly understood at present. Three months of fallowing is generally considered to be a reasonable timeframe and is regularly used. It is currently not clear to what extent sediment recovery occurs under different fallowing regimes or to what degree natural environmental conditions can influence recovery.

The spatial and temporal successional patterns of impact are well defined for temperate systems but there is considerable variability in the timeframe required for recovery. In some cases recovery from fish farming is rapid, taking only a few weeks (Ritz *et al.*, 1989, Brooks *et al.*, 2003), whereas others indicate much longer timeframes (Karakassis *et al.*, 1999, Pereira *et al.*, 2004, Brooks *et al.*, 2004). Level of impact is an important factor affecting recovery level, which in cage aquaculture is affected by farm management criteria (i.e. cage size, stocking density/biomass, feed input and timing/duration of stocked/fallow period). How these factors influence impact/recovery level is also unclear. From the perspective of both farm management and ecosystem protection it is important to have a clear understanding of the processes involved in recovery.

Most studies of sediment recovery associated with fish farming have been restricted to single sites/leases within similar geographic areas (Ritz *et al.*, 1989, Karakassis *et al.*, 1999, Pereira *et al.*, 2004), and there are very few aquaculture-based studies that specifically examine large-scale spatial variability in the sediment recovery response. One exception compared several farm sites and found significant differences in their biological recovery rate, although they all rapidly improved and their chemical

recovery rates were similar (Brooks *et al.*, 2003). In a follow-up study, Brooks et al. (2004) compared the results from the earlier sites with an additional site from the same region and in this case there were significant differences in the biological recovery rates. Recovery at the site in the later study (Brooks *et al.*, 2004) was markedly slower, with the suggestion that it could take more than six years for biological recovery. Although the authors did not specifically examine why this was the case, they suggested that it may be linked to differences in the depositional nature of the particular environment inferring a lack of resilience. An understanding of regional differences in recovery response and the factors underpinning these differences would be extremely useful for management of the local aquaculture industry and organic enrichment in southern temperate areas in general.

What is measured is also extremely important in obtaining a realistic evaluation of recovery. Some measures are much more sensitive to sediment impact/recovery than others. For example, at fish farms in British Columbia, Canada, physical-chemical parameters at cage sites returned to reference conditions within a few weeks, whilst the macrofauna took more than 6 months to recover (Brooks *et al.*, 2003). In Tasmania, the physical and chemical properties of sediments showed that fish farm-derived organic matter levels (identified through fatty acid profiles) remained elevated at cage sites 12 months after the cages were emptied, despite redox potential indicating a return to reference conditions (McGhie *et al.*, 2000). Visual assessment of sediment and epibiotic status is another simple approach which has the potential to provide clear, easily interpretable results. It is often used by regulatory authorities to identify areas of major impact. However, the information obtained is highly subjective. A semi-quantitative approach for video evaluation recently developed by Crawford *et al.* (2001) has increased the value of video assessment. Although sediment impact can be inferred from farm information such as stocking density and feed input, the faunal community structure provides a more integrated indicator of sediment condition. It is well recognised that benthic infaunal evaluation is among the most sensitive of approaches for evaluation of sediment condition.

In most investigations recovery of the sediments was gauged against a return to reference conditions. Total remediation may be appropriate in some cases, for instance

where the disturbance is unique (eg. one-off deposition) and/or unexpected (eg. oil spillages) or where protection of specific rare or important taxa are a concern. In many instance (eg. sewage or pulp mill effluent) there may not be a total removal of impact, only a change in intensity. Management of fish farming recovery represents a very different situation, as in this instance for operations to be sustainable it may not be necessary for sediment condition to return to a reference state, because the impacts are recurrent. Consequently, it may not be useful or necessary to require complete recovery after each production cycle. Recovery to the extent that it does not result in progressive chemical or biological deterioration of sediments may be sufficient to support long-term farming operations. This in turn requires that the level of impact on the sediments does not irreversibly change the ecology of the system. In this situation functional recovery may represent a better benchmark against which to judge recovery. Once the system is functionally restored it should have the capacity to fully recover given sufficient time (Young *et al.*, 2001). In this regard there have been no studies that have investigated appropriate fallowing regimes for such practical management of environmental condition.

Consequently, where the main comparisons are in relation to the system's ability to recover then individual species identities become of less importance and ecological function in relation to successional stage becomes the issue of interest (Rhoads and Germano, 1986). The sustainability of ongoing and repetitive impacts, such as those generated by fish farming may be better assessed by establishing whether the ecological function of the system has been restored. The amount of time actually required for sustainable sediment remediation is at present poorly understood, particularly with respect to southern temperate areas. However, once a system is functionally recovered it will progress to an "equilibrium" community providing no further perturbation is encountered.

1.2 Study Objectives

Understanding the main factors affecting the recovery process is essential to effectively manage the impact of organic enrichment on the environment (Fig. 1.2). Consequently, the primary objectives of this study were to investigate the recovery

response associated with organic enrichment in southern temperate sediments, identifying critical processes and indicators that could be used to improve management practices. The effect of different timescales of recovery (i.e. long- and short-term) on the ecological and geochemical response was of particular interest. Although it is important to understand long-term recovery associated with episodic organic enrichment events such as oil spills or the permanent removal of impacts, there are several important industries, such as caged finfish aquaculture, pulp-mills and sewage treatment plants that generate ongoing or recurrent organic loadings. In these situations information on the recovery response over relatively short time frames is critical for the development of appropriate environmental management strategies.

In a comparison of several aquaculture recovery studies Black (2001) attributed differences in the overall estimates of recovery time to broad scale variability in the underlying environmental conditions. It is recognised that environmental factors become increasingly important as the spatial scale of comparison increases (Zajac *et al.*, 1998). As a result, this study aimed to provide an assessment of the effect of regional variability on the community response and on the extent and rate of recovery. For management of sediment remediation to be effective there needs to be an unambiguous recovery objective as well as a clear understanding of the processes involved. Consequently, in this study I have attempted to define the main factors affecting benthic infaunal recovery in southern temperate soft sediment systems, and examine the importance of these factors in relation to their affects on ecological processes and the level of recovery that can be achieved. Using finfish aquaculture as an example I have attempted to outline the significance of differences in recovery response for environmental management.

1.3 Approach and thesis structure

The complete removal of all cages associated with an Atlantic salmon farming lease provided an opportunity to examine recovery from organic enrichment over the longer term (3 years). The influence of initial impact level on the rate at which the sediments recover was shown by contrasting the recovery response of sediments from directly under the cages with that of sediments in adjacent areas (Chapter 2). The key

ecological changes associated with long term recovery are outlined in chapter 3. In addition alternative approaches for evaluation of recovery are examined and recommendations on the most applicable approaches under the local conditions are discussed (Chapters 2 and 3).

In chapters 4 and 5 the effects of short-term remediation (3 months) are investigated. Regional differences in extent and rate of recovery are described in chapter 4, whilst in chapter 5 the differences in the biological response are more fully investigated and the functional significance of the changes in community composition are discussed.

The management implications of the findings are briefly discussed in each chapter. However, the importance of the study findings for environmental management are examined in more detail in chapter 6, in particular, the ecological significance of functional differences with respect to assessment of recovery and the significance of regional differences for management.

Chapters 2-5 are presented in the form that they were prepared for publication. This has resulted in some repetition of material in introductory and methods sections.

Chapter 2:

Assessment of Long Term Change in Sediment Condition After Organic Enrichment: Defining Recovery

This Chapter previously published as:

Macleod, C.K., Crawford, C.M., and Moltschaniwskyj, N.A. 2004. Assessment of long term change in sediment condition after organic enrichment: defining recovery. Marine Pollution Bulletin 49, 79-88.

Abstract

Sediment condition at an Atlantic salmon (*Salmo salar* L.) culture site in S.E. Tasmania, Australia was evaluated to determine the rate and extent of recovery after removal of farmed fish. By local standards the cage sediment at the start of this survey was markedly degraded but comparison with results from impact studies in Scotland, Canada and Norway suggests that the sediments were considerably less impacted than in northern temperate areas. The impact at the cages diminished rapidly with both time and distance; after only 2 months conditions were markedly improved. The macrobenthos indicated a slower recovery than chemical measures, after 36 months the benthic faunal community structure under the cages still differed from reference conditions even though other sediment measures had recovered. This study highlighted two other key issues in relation to monitoring and management of sediment recovery. First, techniques used to determine impact may not be appropriate for evaluation of recovery. Second, establishment of local baseline standards is extremely important to ensure appropriate evaluation of both impact and recovery.

Keywords: sediment recovery; environmental assessment; monitoring; organic enrichment; aquaculture; video

2.1 Introduction

Deposition of aquaculture waste from finfish cages can result in organic enrichment. To overcome this it is usual for farmers to leave areas of seabed free from farming activities for a period of time to allow recovery. However, it is currently not clear to what extent sediment recovery occurs or to what degree natural environmental conditions can influence recovery. From the perspective of both farm management and ecosystem protection it is important to have a clear understanding of the processes involved in recovery. The degree and extent of organic enrichment of sediments under cages and the magnitude and scale of impact is dependent on both husbandry parameters and physical, chemical and biological characteristics of the environment (Iwama, 1991; Gowen and Rosenthal, 1993; Wu, 1995; Black, 2001). However, few studies have attempted to evaluate sediment recovery rates and results have differed markedly, with estimates of benthic infaunal recovery ranging from 7 weeks (Ritz *et al.*, 1989) to 21 months (Black, 2001) and greater than 23 months (Karakassis *et al.*, 1999). Consequently, the primary objective of this study was to assess the rate of sediment recovery associated with long term fallowing of intensively farmed marine Atlantic salmon cage sites in the temperate waters of south-east Tasmania, Australia.

Many factors influence sediment recovery rate and hence several different techniques have been used as surrogate measures of sediment condition (Hargrave *et al.*, 1997; Kingsford and Battershill, 1998). Some measures (e.g. redox and sulphide) indicate specific chemical aspects of sediment condition (Holmer and Kristensen, 1992; Hargrave *et al.*, 1993) whilst others (e.g. infaunal community structure) reflect a combination of physical, chemical and biological influences (Finlay *et al.*, 1995; Karakassis *et al.*, 1999; Macleod *et al.*, 2002). Characterisation of benthic infaunal communities is one of the most reliable indicators of environmental disturbance, but it can be expensive and time consuming (Wildish *et al.*, 1999; Crawford *et al.*, 2002), consequently simpler and more cost effective techniques are frequently used. These include measurement of redox and sulphide, total organic matter and/or total organic carbon levels and more commonly these days, visual assessment of sediment characteristics and epibiota by diver or video.

Visual assessment of sediment and epibiotic status is a simple approach, which has the potential to provide clear, easily interpretable results. It is often used by regulatory authorities to identify areas of major impact. However, the information obtained is generally subjective. A semi-quantitative approach for video evaluation recently developed by Crawford *et al.* (2001) has increased the value of video assessment.

The second objective of this study was to determine the suitability of different sediment evaluation techniques for assessment of recovery. A number of countries have produced specific protocols for monitoring the impact of fish farms, for example in Norway and Scotland national standards have been developed, whilst regional recommendations exist in Australia and the Canadian provinces. However, although there will be broad similarities in organic enrichment effects, marked geographic differences in the range and scale of measurements, both globally and regionally can be expected. These differences must be taken into consideration when interpreting results among different areas. Furthermore, these guidelines relate specifically to detection of impact, not evaluation of recovery. Although degradation and recovery processes may be similar (Ritz *et al.*, 1989; Karakassis *et al.*, 1999; Black, 2001), it is unlikely that the rates would be equivalent. Sediment recovery is passive whilst degradation results from the active input of waste products. Consequently, the currently acknowledged approaches for evaluating impact may not be appropriate for assessing recovery.

2.2 Methods

2.2.1 Study Site

The study lease (3.12ha) was located on the eastern shore of North-West Bay, Tasmania, Australia (Fig. 2.1). Prior to cessation of farming in August 1999 the farm had been involved in the commercial production of Atlantic salmon (*Salmo salar* L.) for 14 years. Over the preceding four years this site had essentially been stocked continually with little or no fallowing. In the year prior to closure the farm stocked approximately 200-300 tonnes of fish; however stocking levels were markedly reduced in the 3-4 months prior to the site's closure as stock were transferred from the site.

Current velocity throughout the water column was generally slow (3.4–4.3 cms^{-1}), with rates near the seabed further reduced (Macleod *et al.*, 2002).

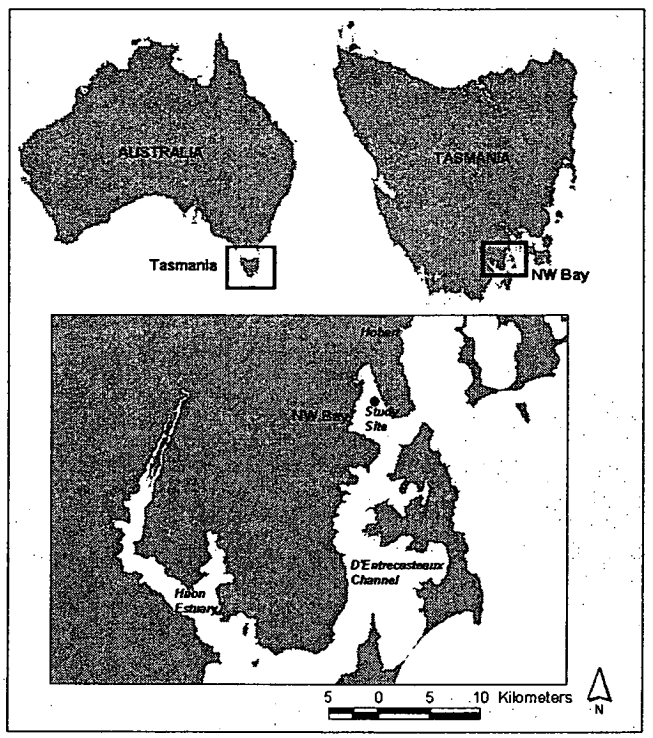


Fig. 2.1 Location of study site.

Two cages were selected at random for this study (Fig. 2.2). These cages had comparable farming histories, i.e. fish biomass and feeding levels were equivalent. At each cage a line on the seabed was run from directly beneath the cage (-10m) to 35m from the cage edge (Fig. 2.2). Stations were established at -10m (under cage), 0m (cage edge), 10m, 20m and 35m. Reference stations for each cage were located 150m from the cages, directly in line with the stations (Fig. 2.2) and at similar depths. The data for each station was averaged across the cages. The positions of 0m, 35m and reference (150m) stations were determined using a differential global positioning system (DGPS).

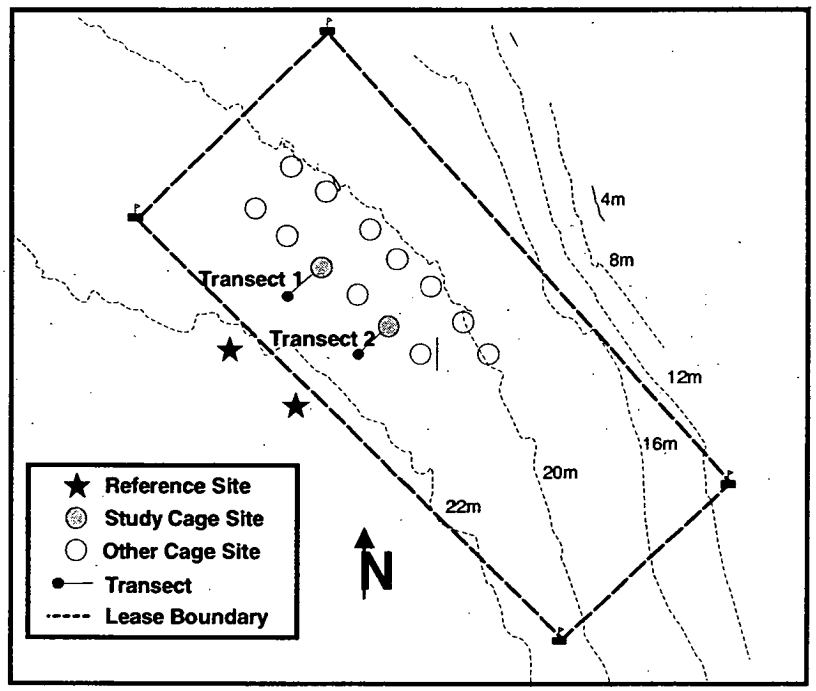


Fig. 2.2 Location of cage study sites and reference stations within study lease.

Initial sampling at each station was undertaken two weeks after removal of fish and then 1, 2, 6, 12, 24 and 36 months thereafter. Benthic samples were collected by diver for macrofaunal and physical/chemical analyses. Video footage was collected along the transect line and from an area within a 2m radius of the reference station.

2.2.2 Physical/Chemical Analyses

At each station three replicate core samples were collected using perspex tubes (250mm long and 45mm internal diameter). A single sub-sample (4ml) was taken from each replicate core at a depth of 4cm for measurement of sulphide using a Cole-Parmer 27502-40 silver/sulfide electrode as per Wildish *et al.* (1999). Sulphide standards were prepared before each sampling event and electrode calibration curves were determined.

After taking samples for sulphide measurements, the remaining sediment was extruded and sectioned. Half of the top 4cm from two cores was collected for sediment particle size analysis. A sub-sample of each was passed wet through a graded series of sieves (4mm, 2mm, 1mm, 500µm, 250µm, 125µm and 63µm). The sediment retained on each sieve was dried and weighed and the percentage of the total sample weight

calculated. The fraction < 63µm was determined as the difference between the initial sample weight and the combined weight of the retained fractions. Total organic matter was determined by the loss on ignition technique (Greiser and Faubel, 1988) modified as follows; samples collected from the top 4 cm of each core were homogenised and a sub-sample of approximately 2-5 g taken, excess carbonate was removed from the samples by 1) sieving to remove large shell fragments and 2) neutralising any remaining carbonate by acidification with 1N HCl. The samples were then oven dried for 24 hours at 60°C before being transferred to a muffle furnace for 4 hours at 500°C. The weight of organic material was calculated as the difference between oven dried and final furnace ashed weights.

2.2.3 Macrofaunal assessment

Macrofaunal data were collected from stations representative of cage impacts (–10m/ 0m), farm effects (10m) and unimpacted reference conditions (150m). At all stations five replicate samples were collected for assessment of the benthic macrofaunal community structure using hand held 150 mm diameter PVC pipe corers to a depth of 100 mm (sampling area of 0.0177 m2). Samples were collected by diver and transferred immediately to mesh bags (0.875 mm2 mesh); on the boat the bags were rinsed and transferred to containers with 40% formalin in seawater. In the laboratory each sample was sieved to 1 mm, sorted and the animals retained were identified to the lowest possible taxonomic level and enumerated.

2.2.4 Video

Video footage was obtained using a Hi-8 underwater colour video camera. Video recordings were assessed at each station and environmental variables were scored as an average value for all frames observed 2 m either side of the stations. Videos were scored according to the criteria described by Crawford *et al.* (2001). The variables measured included a numeric categorisation of sediment colour, *Beggiatoa* density, presence of gas bubbles, feed pellets or farm debris, prevalence of burrows, casts and tracks, abundance of molluscs, ophiuroids, annelids and small fish, and the occurrence of locally common seastar species (*Coscinasterias muricata* and *Asterias*

amurensis).

2.2.5 Statistical Analysis

Univariate data were analysed by Analysis of Variance (ANOVA) with homogeneity of variances checked using residual plots. Data were untransformed. A two-way fixed effects model ANOVA, with factors station and time, was used to assess variation in particle size, organic matter, sulphide concentration and macroinvertebrate diversity (Shannon index, Shannon and Weaver, 1963). Tukey's Honestly Significant Difference post-hoc test was used following a significant ANOVA result.

Multivariate analyses were conducted on the community data and video results using the ecological research software package PRIMER© (PRIMER, 2001). Benthic replicates were combined and square root transformed to adjust the importance of species dominants. Macrofaunal and video data were analysed from 3 positions representative of cage effect (–10m and 0m combined), more general farm effects (10m) and unimpacted conditions (150m).

The data are displayed as ordination plots using non-metric multi dimensional scaling (MDS). Cluster analysis was used to identify groupings within the data and these groupings are identified on the MDS. SIMPER analysis was used to determine if any particular species or factors were indicative of these patterns (Clarke and Warwick, 2001). The interaction among groups and time in the macrofaunal data was evaluated using one-way ANOSIM for the group*time combinations. Where this was significant, pairwise comparisons were made. Video data were assessed using two-way crossed analysis of similarities (ANOSIM) to test for differences in community composition among groups within each time and for differences over time (allowing for the fact that there may be differences between groups). As only a single video assessment was undertaken for each transect there was insufficient power to calculate significance levels for a one-way ANOSIM of all group and time combinations.

2.3 Results

2.3.1 Particle Size Distribution

The predominant sediment type at all stations was silt/clay (<0.063 mm). There was no significant change in the silt/clay fraction at each station over time ($F_{Time*Station}$ =0.859, df=18,40, P=0.625) or through time (F_{Time} =0.549, df=6,40, P=0.766), but there were significant differences between stations ($F_{Station}$ = 24.64, df=5,40, P<0.001). Post-hoc comparisons showed that the proportion of the silt/clay component was significantly lower at the –10 m station than at any of the other stations and was significantly reduced at the 0 m station compared with the reference (Fig. 2.3).

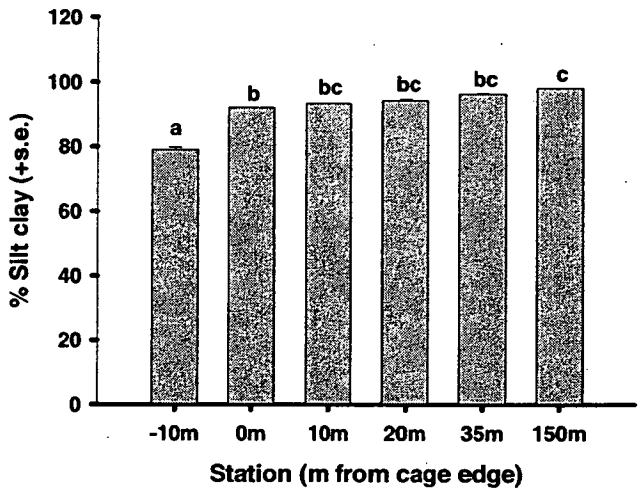


Fig. 2.3 Mean percentage silt/clay (<0.063mm) (+ s.e.) at each sample station averaged across all times (n=54). Where the letters above each bar differ results were significantly different.

2.3.2 Organic Matter Measurement

Organic content was significantly different among stations (F=5.67, df 5,123, P<0.001) and times (F=43.90, df 6,123, P<0.001) but the interaction between station and time was not significant. At the -10 m station organic content was consistently higher than all the other stations, and no other differences were seen (Fig. 2.4a). Initial organic matter levels were high at all stations (c. 20%) (Fig. 2.4b). Levels generally declined by between 30-40% at all stations during the first two years (Fig. 2.4b).

However, in the last twelve months organic matter increased slightly and the overall reduction from the start to end of study was only 10-25%.

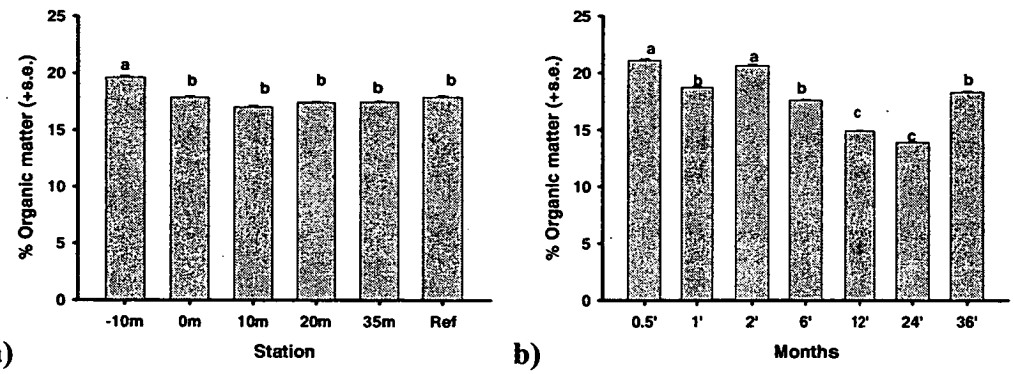


Fig. 2.4 Mean percentage organic matter content (+s.e.) at a) each sample station and averaged across all times b) at each sample time averaged across all stations. The letters above the bars indicate the results of Tukeys post hoc test. Where the letters above each bar differ results were significantly different.

2.3.3 Sulphide

Sediment sulphide levels exhibited a clear spatial and temporal gradation of effect (Fig. 2.5) and a significant interaction between station and time of sampling was identified ($F_{Station*Time}$ 7.244, df=25,17, P<0.001). Sulphide concentration at the cage stations decreased markedly over time. Only in the first 2 months were there significant differences amongst stations (Fig. 2.5). At 1 month the sulphide levels were highest at the –10m station followed by 0m and then 10m stations. Levels were considerably lower at the 20m, 35m and reference (150m) stations and these stations were not significantly different. At 2 months only the –10m station levels were significantly higher (greater than x100) than at the reference stations. Sediment sulphide levels diminished both over time and with distance from the cage site. Levels at the 20 and 35m stations levels remained equivalent to reference throughout the study. Sulphide concentrations at the 0 and 10m stations were similar to the reference within 2 months and by 6 months the -10m stations were comparable to the reference (Fig. 2.5). After 36 months there were no significant differences between any of the stations.

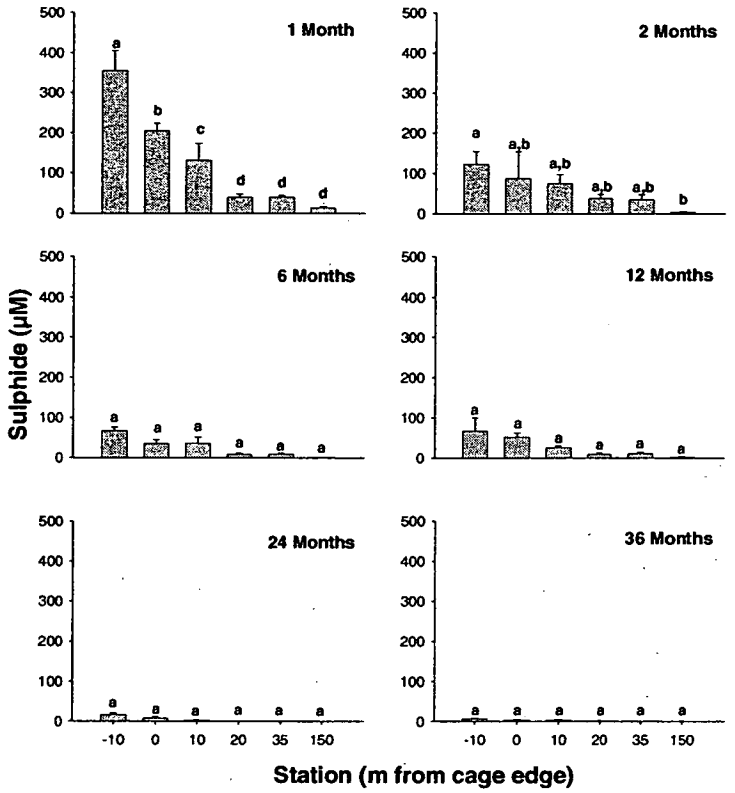
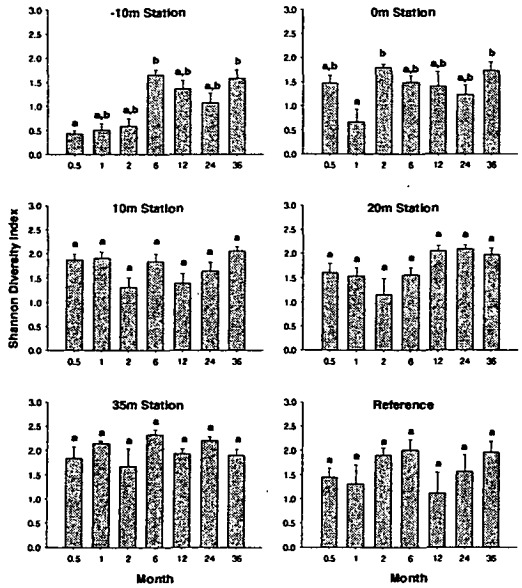


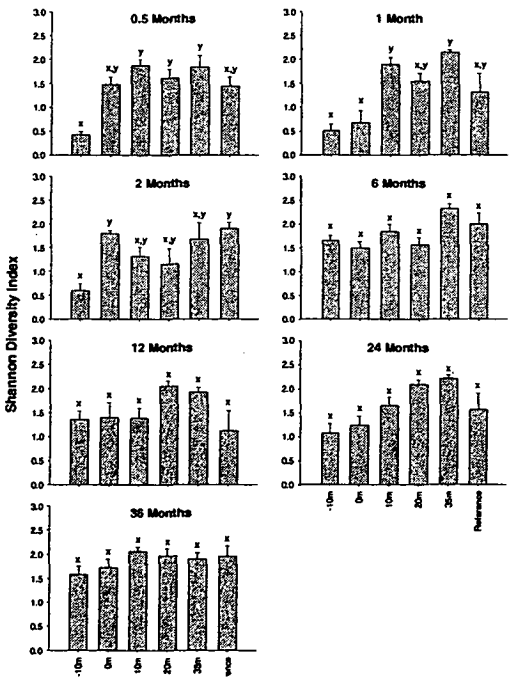
Fig. 2.5 Average sediment sulphide concentration in µM at 4cm depth (+ s.e.) at each sample station and time. (Data not available for 0.5 months). The letters above the bars indicate the results of Tukeys post hoc test, means with different letter are significantly different.

2.3.4 Macrofauna

Changes in diversity, using the Shannon diversity index, indicated an interaction between station and time ($F_{Station*Time} = 2.72$, df=30,210, $P<0.001$). Diversity was consistently > 1.0 at the reference and at stations 10m or more from the cage edge (Fig. 2.6). It dropped to approximately 0.5 at the –10m station for the first 2 months and at the 0m station at 1 month, and only at these stations and times was the diversity significantly lower than the reference (Fig. 2.6). Diversity at the 10m, 20m, 35m and reference stations did not differ significantly over time (Fig. 2.6a) and the diversity at these stations was comparable at each sampling time (Fig. 2.6b).



a)



b)

Fig. 2.6 Average Shannon index value (+ s.e.) at each a) sample station and b) time. The letters above the bars indicate the results of Tukeys post hoc test for comparisons within each station over time (a, b, c) and for all stations within each time (x, y, z), means with different letter are significantly different.

One-way ANOSIM of all group and time combinations for the full community dataset indicated a significant interaction between group and time (Global R=0.454, $p<0.001$) which suggests that the spatial groups responded differently over time. There were no differences between the farm and reference stations through time but the cage communities changed progressively over time (Table 2.1a & b). Except at 24 months the cage and reference communities were significantly different at all equivalent times (Table 2.1a). The cage community at 0.5 months was significantly different to all other times after 1 month (Table 2.1b). A marked change in the community structure was evident after 6 months, after which the cage stations differed significantly from the initial cage community. By 36 months the cage stations were different from all times earlier in the study, and although not significantly different from the farm station, they were still significantly different from the reference station (Table 2.1b). The only difference between the farm and reference communities was at 0.5 months (Table 2.1b).

Table 2.1 Probability values from the one-Way ANOSIM comparison of benthic data for selected sample station groups (C-cage and R-reference) and times (Bonferroni corrected n=84, $p<0.0006$). a) comparison of cage and reference communities within times, b) comparison of cage communities over time. Categories with significant differences are shown in bold. The only significant difference between the farm and cage communities was at 0.5 months (p=0.0002) and there were no significant differences between the farm and reference communities, so farm results not shown.

a)

Cage vs Reference at each time	Probability
0.5 month	**0.0001**
1 month	**0.0006**
2 months	**0.0002**
6 months	**0.0003**
12 months	**0.0002**
24 months	0.003
36 months	**0.0001**

b)

Group/	0.5C	1C	2C	6C	12C	24C
1C	0.2219					
2C	**0.0023**	0.1845				
6C	**0.0001**	**0.0006**	**0.0005**			
12C	**0.0001**	0.0037	0.0062	0.0043		
24C	**0.0001**	0.0015	**0.0006**	0.0028	0.0098	
36C	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**

The two dimensional ordination plot (Fig. 2.7) shows a spatial progression with the cage stations on the far left of the plot, the reference stations on the far right of the plot and the farm stations forming a central group. At the cage stations a temporal gradation was also evident within the spatial distribution; the earliest impacted samples tending towards the left and the later samples tending towards the right. The demarcations between the cage and farm groups and between the farm and reference groups are not well defined indicating that both the temporal and spatial changes in community structure were gradual rather than sudden. However, the differentiation of the cage stations from the reference stations was clear.

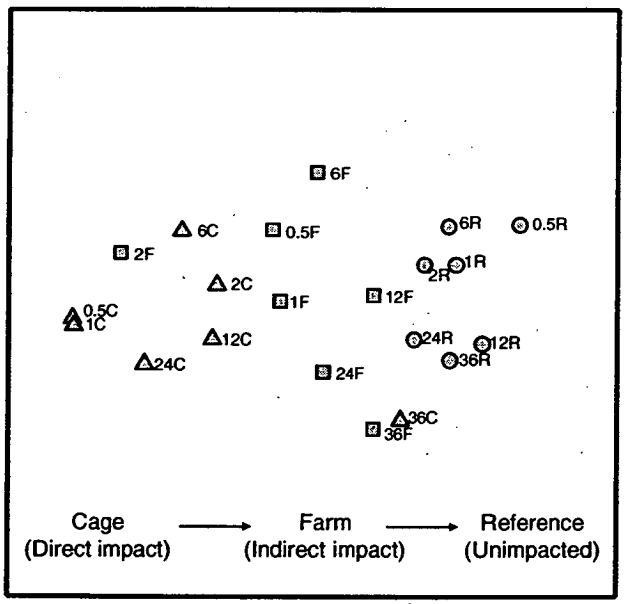


Fig. 2.7 Ordination analysis – 2-dimensional MDS plot of species abundance data. Stress=0.12. The prefix indicates the time of sampling in months whilst the symbol and suffix indicate the impact group, cage (▲), farm (■) and reference (●) respectively.

2.3.5 Video Assessment

Two-way ANOSIM of the *a priori* groups (cage, farm and reference) and time indicated significant differences between both groups (Global R=0.326, P<0.001) and times (Global R=0.288, P<0.001). Pairwise comparisons showed that within groups

there were significant differences between the cage and all other groups (Cage/Farm R=0.250, p=0.002; Cage/Reference R=0.511, p=0.001) and also between the farm and reference groups (Farm/Reference R=0.418, p=0.001). Pairwise comparisons of all time combinations, adjusted for multiple comparisons (n=21, p=0.002), indicated that the visual condition of the sediments at 0.5 months was significantly different from that at 12 months. Video footage at 2 months differed from that at 12 and 24 months and footage from 6 months was significantly different to that from 24 months. However, differences between communities at the later sample times (12, 24, 36 months) were less obvious. Ordination of the video assessment data reveals a spatial gradient in the stations across the plot which largely separates the cage stations in the first 12 months from the remaining stations (Fig. 2.8). These remaining stations were statistically indistinguishable from one another. SIMPER analysis (Table 2.2) of the two main groups identified profusion of burrows and faunal tracks, sediment colour and the presence of *Beggiatoa* as the primary factors in the group determination.

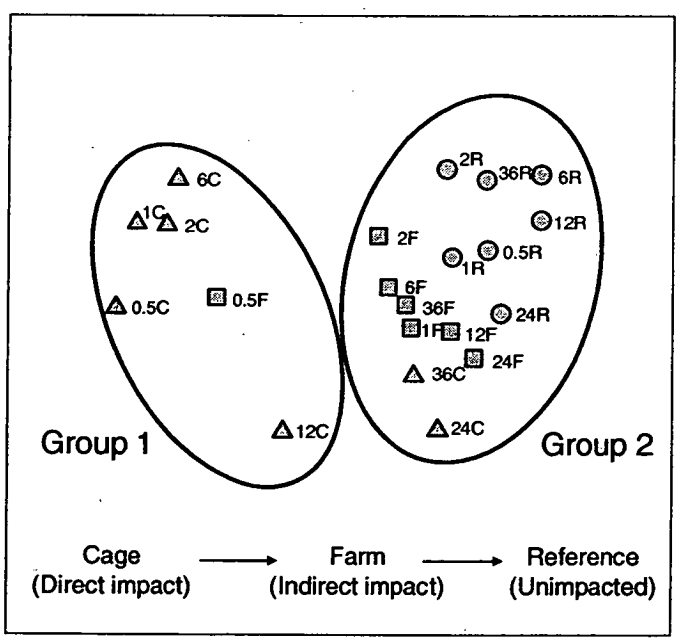


Fig. 2.8 Ordination analysis – 2-dimensional MDS plot of video assessment data from all sample stations where video footage was usable. Stress=0.09. The two primary groups identified by cluster analysis at a similarity level of 44% are shown. The prefix indicates the time of sampling and the symbol and suffix indicate the impact group,

cage (⚠), farm (▣) and reference (◎) respectively.

Table 2.2 SIMPER output for the video assessment indicating a) and b) average abundance, ratio (average similarity / st.dev. similarity), % similarity and cumulative % similarity of the most important variables in each of the *a priori* groups (Cage, Farm and Reference) and c) average abundance, ratio (average similarity / standard deviation similarity) and cumulative % similarity of the five variables which most clearly distinguish the main groups identified by cluster analysis.

Species Name	Average abundance	Ratio	Percent Similarity	Cumulative % Similarity
a) GROUP 1				
Sediment colour	1.43	2.21	42.09	42.09
Beggiatoa density	1.10	1.11	22.80	64.89
Worm cast density	0.57	1.27	13.98	78.87
b) GROUP 2				
Burrow density	2.21	3.72	39.50	39.50
Density of faunal tracks	1.48	1.86	23.47	62.97
Mollusc abundance	0.96	2.10	14.51	77.48

Species Name	Group 2 Average Abundance	Group 1 Average Abundance	Ratio	Cumulative % Similarity
c)BETWEEN GROUPS				
Burrow density	2.21	0.33	2.95	18.72
Sediment colour	0.06	1.43	2.34	32.87
Density of faunal tracks	1.48	0.10	2.20	46.85
Beggiatoa density	0.00	1.10	1.52	57.82
Mollusc abundance	0.96	0.05	1.96	67.01

2.4 Discussion

The high levels of silt and clay in the sediment of the study site signify that both within the lease area and at the reference stations the sediments were "depositional" (Rosenthal *et al.*, 1988) and the flow rates indicate that the extent of waste dispersion from the cages would be limited. This suggests that the benthic impacts from aquaculture operations would be highly localised but would also not be readily mitigated by natural hydrographical processes. Fish farm sediments generally have a very high organic matter content which can be strongly anoxic and rich in sulphides (Brown *et al.*, 1987; Frogh and Schanning, 1991; Brooks *et al.*, 2003). Recent

regulations proposed for British Columbia, Canada (Levings *et al.*, 2002) identify the "trigger" standard for sulphide as 1,300 μM, whilst Scottish regulations set their minimum limit for action at 3,800mg/kg sediment dry wt (~20,000μM) (SEPA, 1998). Under these guidelines even the most impacted samples from the current study would fall well within the acceptable range. Using the sulphide level categorisation proposed by Wildish *et al.* (1999) for New Brunswick, Canada, the highest sulphide levels observed in the present study indicate only hypoxic/ moderately polluted conditions. The sulphide levels at all stations also diminished rapidly, suggesting that the sediments were recovering and after 24 months levels were indistinguishable from background conditions.

Measurement of organic matter has been widely used as a surrogate for organic enrichment. However, several recent studies suggest that evaluation of organic matter content is not always a useful measure of farm impact (eg. Johannessen *et al.*, 1994; Hargrave *et al.*, 1997; CSIRO Huon Estuary Study Team, 2000; Macleod, 2000; Crawford *et al.*, 2002; Brooks *et al.*, 2003). In the present study organic matter levels recorded from all samples were very high, (~20% in association with cages and ~18% at the reference stations). Levels directly under the cages were generally higher than those reported from farming operations under similar environmental conditions either overseas, 9.5% (Brown *et al.*, 1987), or locally 16-17% (Macleod, 2000). However, in the recent study of the nearby Huon estuary comparably high levels (18-24%) were observed at stations in the upper reaches of the estuary where the input of terrestrial organic material was significant (CSIRO Huon Estuary Study Team, 2000). Results for bulk organic matter parameters in the Huon estuary indicated that a significant portion of organic waste remained in the sediments after twelve months and suggested the residual organic material may be more refractory (McGhie *et al.*, 2000) and therefore not as readily available to the biota. Organic matter levels directly under the cages remained high throughout this study, whereas other aspects of the sediment chemistry and biology indicated improvements in sediment conditions. It may be that a large proportion of the organic matter is refractory and therefore is not assimilated. Nonetheless, the results suggest that measurement of organic matter level is a poor indicator of sediment recovery.

At the time of cage removal the macrobenthic community structures at the cage associated stations (-10 m and 0 m) were clearly impacted. The fauna was impoverished, species diversity (Shannon index) was very low, and the community structure was similar to that described by Pearson and Rosenberg (1978) as characterising the "polluted" zone. The community was dominated by the opportunistic polychaete *Capitella capitata* complex, a species indicative of organically enriched conditions. Multivariate analysis of the data clearly indicated that the community structure of these stations changed over time in a manner suggestive of recovery. Nevertheless, the community structure at the under cage stations (-10m) remained impacted 36 months after the cages had been removed and a moderate impact could still be distinguished at the 0m stations after 24 months. At the farm stations it was difficult to discern a clear impact at any time, although the community often contained transitional species.

Estimates of benthic infaunal recovery from caged fish farming have ranged from 7 weeks in coastal waters off S.E. Tasmania (Ritz *et al.*, 1989) to 21 months from the west of Scotland (Black, 2001) and greater than 23 months in relation to sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*) culture in Cephalonia Bay, Greece (Karakassis *et al.*, 1999). The variability in these earlier estimates is probably a combination of both ecosystem and farm management differences. The studies by Black (2001) and Karakassis *et al.* (1999) are more comparable to the current study even though the culture species differ. Black (2001) contrasted his results with those of Karakassis *et al.* (1999) and noted that the recovery rates seemed to be much higher in the warmer waters. However, local hydrographic conditions influence recovery rates and in quiescent areas recovery may take much longer than in more hydrodynamically energetic areas (Black 2001). The difference between the rate of recovery observed in the current study and that of the earlier study by Ritz *et al.* (1989) is probably in part due to differences in background environmental conditions and in part to the expansion and intensification of the industry since that time. There have been significant changes in cage design and stocking densities are now much greater.

Although the benthic community changes observed in the current study were similar to those described elsewhere (Johannessen *et al.*, 1994; Findlay *et al.*, 1995;

Karakassis *et al.*, 1999; Wildish *et al.*, 1999; Brooks *et al.*, 2003), there were marked differences between the absolute quantities of the chemical indicators and the level of impact inferred by the changes in community structure. Overall trends were similar, with both sulphide and organic matter levels increasing significantly where major community changes occurred. However, the magnitude of change was very different - sulphide levels were substantially lower under Tasmanian conditions than observed in Canada or Scotland and organic matter levels were considerably higher (Gowen *et al.*, 1988; Holmer and Kristensen, 1992; Wildish *et al.*, 1999; Brooks *et al.*, 2003). This clearly reflects geographical differences and suggests that although trends may be similar, absolute levels cannot be directly extrapolated over large spatial scales.

Video is regularly employed by farmers in Tasmania as a means to examine and evaluate seabed condition. Video footage is generally assessed qualitatively and the current study identified several characteristics which represented consistent indicators of severe impact. Bacterial mats (*Beggiatoa* spp), blackened sediments and gas bubbles are clear visual indicators of impacted sediments (Crawford *et al.*, 2001). The video assessment suggested that the density (size and thickness) of the *Beggiatoa* mats had increased at -10m stations between 0.5 and 1 month after removal of the cages. The presence of *Beggiatoa* mats at the –10m stations as late as 6 months after cage removal suggests that the sediment was still anoxic. *Beggiatoa* mats develop at the interface between hypoxic and anoxic conditions, requiring the presence of both sulphide and oxygen (Frogh and Schanning, 1991). Consequently diver observations of no *Beggiatoa* under the cages at initial sampling and subsequent increase in mat density in the first month, suggested that initially the sediment was anoxic and *Beggiatoa* development was inhibited. The presence of infaunal species under these conditions is probably a function of the particular species capabilities. These species were generally highly tolerant of hypoxia, and were able to irrigate their burrows by extending tubes into the better oxygenated overlying water.

Assigning values to observed video features allowed direct comparison between locations and over time. The multivariate analysis of the video parameters in the current study suggested that at 10m from the cage the sediment had recovered sufficiently to be indistinguishable from the reference conditions after only 1 month,

but that there was still a significant impact beneath the cages 12 months after their removal. This approach for evaluation of video footage is relatively simple and makes information obtained from video footage more useful. Video data is relatively quick and easy to collect and video results can assist the interpretation and presentation of data produced by other measurement techniques. Video footage is also extremely effective in presenting highly impacted conditions.

This study showed marked differences in the sensitivity of a number of different assessment techniques. Sediment chemistry responded to the changing environmental conditions more quickly than the benthic infaunal community. Video assessment was an effective means of evaluating recovery, although it also indicated a more rapid recovery than the benthic community.

Although the pattern of recovery indicated by the macrofauna in the present study was consistent with that reported from the northern hemisphere (Pearson and Rosenberg,1978; Johannessen *et al.*, 1994; Findlay *et al.*, 1995; Karakassis *et al.*, 1999; Wildish *et al.*, 1999; Brooks *et al.*, 2003) geographic differences between the levels of impact suggested by associated sediment chemistry measurements were apparent. This highlights the importance of collecting baseline information and determining sediment recovery/degradation rates in relation to local environmental conditions when environmental regulations/guidelines are being established.

Chapter 3:

Ecological and Functional Changes Associated with Long-Term Recovery from Organic Enrichment

This Chapter submitted for publication to Marine Ecology Progress Series as:

Macleod, C.K., Moltschaniwskyj, N.A. and Crawford, C.M. Ecological and Functional Changes Associated with Long-Term Recovery from Organic Enrichment.

Abstract

The recovery of a soft sediment benthic invertebrate community from high levels of organic enrichment was evaluated after removal of farmed fish at an Atlantic salmon (*Salmo salar*) culture site in S.E. Tasmania. Although the pattern of recovery followed established successional principles, after 36 months neither communities under or at the edge of cages, nor farm communities 10 m from the edge of the cage had attained a structure equal to that of the reference communities. In the first few months there was little evidence of recovery and the community was dominated by species characteristic of impacted conditions. After 12 months the system had markedly improved, containing a diverse range of species with functional roles similar to that of the reference sites. Once the ecological function of the sediment was restored subsequent changes in the community structure were relatively minor, simply reflecting the addition of rarer climax species with longer reproductive cycles and/or larval stages with a greater sensitivity to the negative effects of organic enrichment. These species may be significant where specific community changes, loss of diversity or the possibility of species extinction are the critical issues however, restoration of system function may be a more useful indicator of generalized recovery from organic enrichment than community equivalence. The main limitation to such assessments is the lack of biological and ecological information on Tasmanian species.

Keywords: Tasmania, organic enrichment, recovery, functional ecology, aquaculture, benthic infauna

3.1 Introduction

The spatial and temporal successional patterns of marine benthic recovery from organic enrichment are well defined (Pearson and Rosenberg, 1978, Rhoads *et al.*, 1978). However, there is considerable variability in estimates of the timeframe required for recovery. Recovery from organic enrichment resulting from fish farming can be fairly rapid, taking only a few weeks (Ritz *et al.*, 1989, Brooks *et al.*, 2003), or it can occur over longer timeframes (Karakassis *et al.*, 1999, Brooks *et al.*, 2004). The ways in which recovery is defined and assessed may account for some of the differences in these estimated times of recovery. Geochemical recovery of the sediments is generally more rapid than biological recovery (e.g. Carroll *et al.*, 2003, Chapter 2). Recovery of species diversity or faunal abundance, restoration of particular species or community types or full community equivalence with unimpacted conditions are all valid criteria for the determination of recovery but have very different recovery timeframes, with full community recovery likely to provide the most conservative estimates. Knowing when particular components of the community re-establish may be an important measure of recovery where biodiversity is the main issue, or where rare or endangered species are a particular concern. Consequently, when comparing recovery rates it is very important to ensure that the underlying measure is equivalent. When making management decisions regarding recovery it is important to be clear on the level of recovery upon which those decisions are being made and to establish "a priori" what is meant by recovery.

The classic studies by Pearson and Rosenberg (1978) and Rhoads *et al.* (1978) define temporal and spatial successional patterns associated with organic enrichment. These models have been validated in many subsequent investigations for a variety of organic enrichment sources, including aquaculture (e.g. Weston, 1990; Holmer and Kristensen, 1992; Findlay *et al.*, 1995; Hargrave, *et al.*, 1997, Karakassis *et al.*, 1999). These models identify "indicator" species or community types that reflect particular successional stages of disturbance and recovery. The identification of these "indicator" species and their role in the recovery process remains a cornerstone of both basic and applied studies of benthic recolonisation (Whitlach *et al.*, 2001). However, with the

exception of the opportunistic species *Capitella capitata*, these indicator species are not well known in southern temperate regions.

In Tasmania the overall trends in response to organic enrichment are similar to those observed in the northern hemisphere, but there is a marked difference between the absolute levels of the chemical indicators and the extent of impact inferred by the changes in the community structure (Chapter 2). The findings from Chapter 2 indicate that the biological response of the Tasmanian fauna was much greater for a lesser chemical impact than observed in the northern hemisphere, suggesting that geographic location can have a significant influence on recovery response. However, this study did not specifically examine the faunal relationships and so did not distinguish specific ecological changes associated with the key stages in recovery.

Many other factors can also influence the recovery response. Clearly the nature, frequency and size of the impact will all have important consequences for subsequent recovery (Zajac *et al.*, 1998). Hydrodynamic condition is also a critical factor, and will have a major influence on recovery rate (Black, 2001). Local sediment conditions will significantly affect the benthic ecology, determining the community composition and the systems' capacity to respond to environmental stressors (Snelgrove and Butman, 1994). The resilience of the background community is of particular importance in determining recovery response (Bonsdorff, 1989). Some early colonizing "opportunistic" species have life history characteristics that facilitate rapid responses in recently perturbed areas, while late successional "equilibrium" species are more constrained in their population responses and have much slower recolonisation rates (Whitlach *et al.*, 2001). Species interactions may facilitate or inhibit successional changes (Connell and Slatyer, 1977). All of these interactions can be affected by the timing of the disturbance event.

The amount of time actually required for sustainable sediment remediation is at present poorly understood. However, if we accept the assertion that once a system is functionally recovered it will progress to an "equilibrium" community providing no further perturbation is encountered (Young *et al.*, 2001) then remediation should be sustainable once sediment function is restored. Consequently, individual species

identities become of less importance and the ecological function of the species in relation to the particular successional stage with which it is associated becomes the issue of interest (Rhoads and Germano, 1986). This in turn suggests that the sustainability of ongoing and repetitive impacts, such as those generated by fish farming may be better assessed by establishing whether the ecological function of the system has been restored.

This study is a continuation of previous work (Chapter 2) examining long-term changes in sediment condition after the cessation of organic enrichment and focuses on the biological recovery process. The main aims of this study were to improve our understanding of local ecological changes by identifying the resident taxa associated with the critical successional stages and to define the functional significance of those species in the recolonisation/ successional process.

3.2 Methods

The location and farming history of the study area and sampling stations are described in detail in chapter 2. Two cage stations with comparable farming histories, i.e. equivalent fish biomass and feeding levels, were selected at random for this study. At each cage sampling stations were established at -10m (centre cage), 0m (cage edge) and at 10m. Reference stations for each transect were located 150m from the cages, directly in line with the fixed transects and at similar depths. The positions of all stations were established using a differential global positioning system (DGPS).

Benthic samples were collected by diver at each station two weeks after removal of fish and then 1, 2, 12, 24 and 36 months after removal. At each station five replicate benthic samples were collected using hand held 150 mm diameter PVC pipe corers to a depth of 100 mm (sampling area of 0.0177 m^2). Samples were transferred immediately to mesh bags (0.875 mm^2 mesh); on the boat the bags were rinsed and transferred to containers with 40% formalin in seawater. In the laboratory each sample was sieved to 1 mm, sorted and the animals retained were identified to the lowest possible taxonomic level and enumerated. Samples were grouped as representative of cage impacts (–10m/ 0m), farm effects (10m) and unimpacted reference conditions (150m). The data

analysed in this chapter represent a subset of the data used in chapter 2.

Univariate statistical analyses were used to examine the spatial and temporal differences in the community structure. Analysis of variance (ANOVA) was used to evaluate differences in abundance both between sampling stations and within stations over time.

Multivariate techniques were used to determine the relationships between the community structures with varying distances from the cage. Accordingly the samples taken at each station (cage, farm, and reference station) for both of the study cages were combined giving a total of 10 replicates per station. The data were square root transformed to adjust the importance of species dominants and a similarity matrix was derived from abundance data using Bray-Curtis similarity indices. The data are displayed as ordination plots using non-metric multidimensional scaling (MDS). The significance of differences in the macrofaunal data between the stations over time was evaluated using planned comparisons in the ANOSIM randomization test. SIMPER analysis was used to determine if any particular species were indicative of these patterns. Multivariate analyses of community data was undertaken using the ecological research software package PRIMER© (PRIMER, 2006).

Key ecological and reproductive strategies were defined for the main species determined by SIMPER analysis to define the community groups. The fauna was broadly characterised and compared in relation to their general ecological function. The main ecological and life history groupings were feeding strategy, function in sediment, and reproductive strategy. Three main feeding strategies were identified: predatory carnivores (C), suspension feeders (SF), or deposit feeders (DF). However, many animals change their feeding strategies in response to environmental conditions, in some cases this may be a well known and specific change (i.e. DF/C) or the changing strategies may be unknown VF (variable feeding strategy)). Whether the animals were sediment bioturbators (DS) or consolidators (ST) was defined by reference to the known ecology of the species or of that of the nearest relative. The main reproductive strategy of each species was characterised by whether adults had staged (SR) or opportunistic (OR) reproductive cycles and whether larvae were benthic (B) or pelagic

(P). Australian marine and estuarine infauna is relatively poorly described, with very little species specific biological and ecological information. Where the ecology/biology is not specifically known it was inferred using information from the next closest species or taxonomic group. Comparisons were made between the communities to define the ecological significance of the changes. Principal components analysis (PCA) of the ecological data was used to reveal differences between cage, farm and reference stations. The first two principal components (PC) accounted for greater that 95% of the overall variation consequently inclusion of higher level PCs was deemed unnecessary. The associated biplot shows the functional groups and reproductive strategies most responsible for the community changes.

3.3 Results

There was no significant change in the reference communities over time (all P values < 0.0008; corrected for multiple comparisons) (Fig. 3.1). Three species, *Lysilla jennacubinae*, *Nucula pusilla* and *Amphiura elandiformis* were consistently abundant in the reference communities over time (Fig. 3.2, Table 3.1). With the exception of 24 months sample, these species accounted for >50% of the overall similarity at the reference stations (Table 3.1c). The community at the cage station at 36 months was more similar to that of the reference sites than at any other time (Fig. 3.1). The difference between the cage community at 36 months and those of earlier sample times was largely a result of an increase in the numbers of *Paraprionospio coora*, a decline in the abundance of *Euphilomedes* sp.1 and the absence of Nemertea sp.1, *Simplisetia amphidonta*, *Malacoceros tripartitus* and *Capitella capitata* complex.

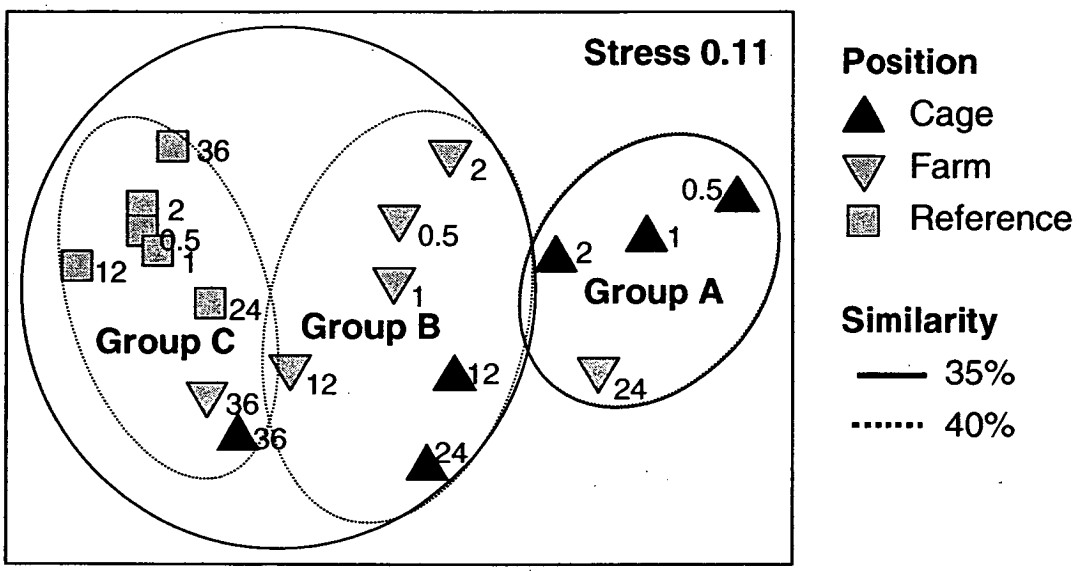


Fig. 3.1 MDS ordination of relationships between benthic faunas sampled at cage, farm and reference stations over 36 months. Numbers indicate the month that samples were taken. Cluster groups identified at 40% similarity level are indicated.

Although there was no difference in the average number of species recovered over time (ANOVA F= 1.04, df 2, 157, p= 0.394), there were significantly fewer species per sample at the cage stations than at either the reference sites or the farm stations (ANOVA F= 3.80, df 2, 157, p= 0.025) (Table 3.2). There was a significant interaction between time and station with respect to mean abundance (ANOVA F= 2.355, df 10, 157, p= 0.013), such that there were greater numbers of individuals at the cage station at 0.5 months (Table 3.2). A substantial number of reference species were present at the cage and farm stations from the onset of fallowing. After 12 months the proportion of reference species at the cage and farm stations was 53% and 67% respectively, but interestingly these proportions declined markedly in subsequent samples (Table 3.2).

Cluster analysis and ordination of the community data revealed three community groups at an overall similarity level of 40% (Fig. 3.1). The most impacted community group (Group A) comprised the initial cage stations (0.5-2 months) and the farm station at 24 months. This community was strongly dominated and therefore characterised by *Capitella capitata* complex and *Malacoceros tripartitus*, which together accounted for

63% of the overall group similarity (Table 3.3). The intermediate community (Group B, containing most of the farm stations and the cage stations at 12 and 24 months) was characterised by *Theora fragilis* and *Euphilomedes* sp.1, which together accounted for 50% of the overall group similarity (Table 3.3).

The primary difference between the cage stations in the initial few months (group A) and the farm stations in the intermediate impact group (group B) was a markedly lower abundance of *Capitella capitata* complex at the farm stations (Fig. 3.2). Impact related species were largely confined to the cages stations over the first two years (Fig. 3.2). After 12 months *C. capitata* complex and *Malacoceros tripartitus* were found only occasionally, and in low numbers, at either the farm or reference stations, although there was an increase in the abundance of *C. capitata* complex at the farm station at 24 months. The "clean" indicator species *Lysilla jennacubinae* was consistently abundant at the reference sites. It was well established at the farm and cage stations at 36 months, but only occurred infrequently, and in low numbers, at other times (Fig. 3.2). Similarly both *Nucula pusilla* and *Amphiura elandiformis* were also regularly present at the reference stations but only occurred occasionally at other stations. *Lumbrinereis* sp. was also found at the reference stations at all times and at the farm stations from 12 months, with a small number at the cage and farm stations at 1 month. *Euphilomedes* sp.1 was relatively ubiquitous, although the greatest abundances were at the farm station at 1 and 12 months and the cage station at 12 and 24 months, i.e. mid-way through the recovery process (Fig. 3.2). The introduced species *Corbula gibba* and *Theora fragilis* were relatively common in both the impacted and reference communities.

Table 3.1 Reference group a) average abundance (no/m2), similarity levels and contribution to group similarity, b) average abundance of characterising species over time and c) % contribution to group similarity of characterising species over time.

a) Reference Group Average similarity: 28.79

Species	Av.Abund	Av.Sim	Contrib%
Lysilla jennacubinae	49	6.72	23.33
Nucula pusilla	64	6.39	22.19
Amphiura elandiformis	51	4.74	16.46
Euphilomedes sp.1	20	2.08	7.22
Lumbrinereis sp.1	14	1.69	5.88

b) Average Abundance (no/m2)

Species	0.5	1	2	12	24	36
Amphiura elandiformis	57	33	48	18	18	152
Nucula pusilla	61	67	91	31	42	53
Lysilla jennacubinae	21	60	107	21	12	209
Euphilomedes sp.1	19	31	22	9	56	0
Corbula gibba	17	20	8	25	9	9
Theora fragilis	5	11	2	25	9	9
Lumbrinereis sp.1	11	7	31	9	56	0
Nemertea sp.1	10	11	23	25	9	9
Mediomastus australiensis	0	0	0	27	0	0
Phoronida sp.1	3	0	2	31	42	30

c) %Contribution to Group Similarity

Species	0.5	1	2	12	24	36
Amphiura elandiformis	29	12	13	3	2	33
Nucula pusilla	23	24	25	8	5	28
Lysilla jennacubinae	9	31	29	53	24	4
Euphilomedes sp.1	8	8	3	2	34	0
Corbula gibba	8	4	1	<1	2	<1
Theora fragilis	2	5	<1	2	8	19
Lumbrinereis sp.1	3	2	10	21	2	<1
Nemertea sp.1	3	4	6	0	3	0
Mediomastus australiensis	0	0	0	8	0	0
Phoronida sp.1	<1	0	<1	<1	0	4

Table 3.2 Mean number of species (± s.e.), mean abundance (± s.e.) and total number of species at each station and time and percentage of species common to references.

Station	Mean Number of Species (± s.e.)	Mean Abundance (± s.e.)	Total Number of Species	% Species Common to Reference
Cage				
0.5	5 (± 15.4)	21,146 (± 4,510)	25	45%
1	4 (± 13.0)	8,521 (± 2,503)	20	28%
2	6 (± 12.7)	10,065 (± 2,399)	36	65%
12	6 (± 12.1)	5,581 (± 1,709)	25	53%
24	5 (± 16.7)	4,316 (± 2,374)	13	33%
36	7 (± 8.1)	3,740 (± 880)	16	38%
Farm				
0.5	9 (± 8.5)	8,303 (± 1,483)	29	64%
1	9 (± 13.5)	9,516 (± 2,409)	35	47%
2	5 (± 14.9)	5,107 (± 2,321)	22	35%
12	6 (± 17.2)	4,551 (± 2,327)	17	67%
24	6 (± 33.1)	15,698 (± 8,106)	13	29%
36	6 (± 10.6)	3,440 (± 1,300)	17	38%
Reference				
0.5	7 (± 9.0)	4,158 (± 1,201)	22	
1	7 (± 16.7)	7,156 (± 2,973)	32	
2	7 (± 14.1)	6,686 (± 2,861)	23	
12	5 (± 18.8)	3,684 (± 2,517)	15	
24	7 (± 21.1)	6,273 (± 2,824)	21	
36	10 (± 16.9)	10,960 (± 3,321)	29	

The functional ecology of the dominant species (i.e. those comprising more than 90% of the overall similarity) within the impacted (cages 0.5-2 months) and reference communities provide important information on the ecological significance of the community changes (Table 3.4). The reference communities contained species with a diverse range of ecological and reproductive functions. In contrast there was markedly less functional variability in the impacted communities. There were fewer sediment stabilisers or suspension feeders, both in terms of abundance and number of species, in the impacted communities, but the proportion of taxa (species and abundance) with opportunistic reproductive strategies was much greater. Nemerteans and echinoderms were important taxa in the reference communities but played little part in the communities at the impacted sites, where the species mix and abundance was dominated by annelids.

Table 3.3 Average abundance (No m^{-2}), average similarity, similarity ratio and contribution to group similarity of the characterising species (i.e. species comprising 90% of the within group similarity for each of the cluster groups distinguishable at greater than 40% overall similarity) for each community group.

Group A *(0.5-C,1-C, 2-C, 24-F)* Average similarity: 60.52

Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**
Capitella capitata	1701	32.18	8.19	53.17
Malacoceros tripartitus	140	6.15	1.25	10.17
Theora fragilis	29	3.61	1.65	5.96
Prionospio kulin	25	3.36	0.90	5.56
Euphilomedes sp.1	31	3.27	1.42	5.40
Corbula gibba	18	2.64	8.31	4.36
Nassarius nigellus	18	2.43	5.06	4.02
Simplisetia amphidonta	11	2.43	10.62	4.02

Group B *(0.5-F,1-F,2-F,12-C,24-C,12-F)* Average similarity: 47.29

Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**
Theora fragilis	121	12.95	2.37	27.39
Euphilomedes sp.1	96	10.96	1.94	23.18
Corbula gibba	19	3.55	1.47	7.51
Nassarius nigellus	21	3.43	1.22	7.25
Nemertea sp.1	9	2.77	1.29	5.86
Prionospio kulin	15	2.49	0.87	5.27
Simplisetia amphidonta	10	2.06	0.76	4.36
Capitella capitata	9	2.04	0.97	4.32
Amphiura elandiformis	4	1.10	0.59	2.32
Phoronida sp.1	2	0.74	0.69	1.56
Mysella donaciformis	2	0.68	0.48	1.43

Group C *(0.5-R,1-R,2-R,12-R,24-R,36-R,36-F,36-C)* Average similarity: 50.80

Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**
Lysilla jennacubinae	34	6.82	2.59	13.43
Euphilomedes sp.1	24	5.75	2.08	11.31
Nassarius nigellus	20	5.22	2.29	10.28
Theora fragilis	25	4.83	1.89	9.51
Nucula pusilla	34	4.65	0.98	9.16
Amphiura elandiformis	30	3.97	0.99	7.82
Corbula gibba	9	3.87	5.44	7.61
Lumbrinereis sp.1	11	3.64	1.40	7.17
Euphilomedes sp.2	7	3.42	2.77	6.74
Euchone limnicola	4	2.06	1.99	4.05
Chaetozone setosa	4	1.56	0.93	3.07

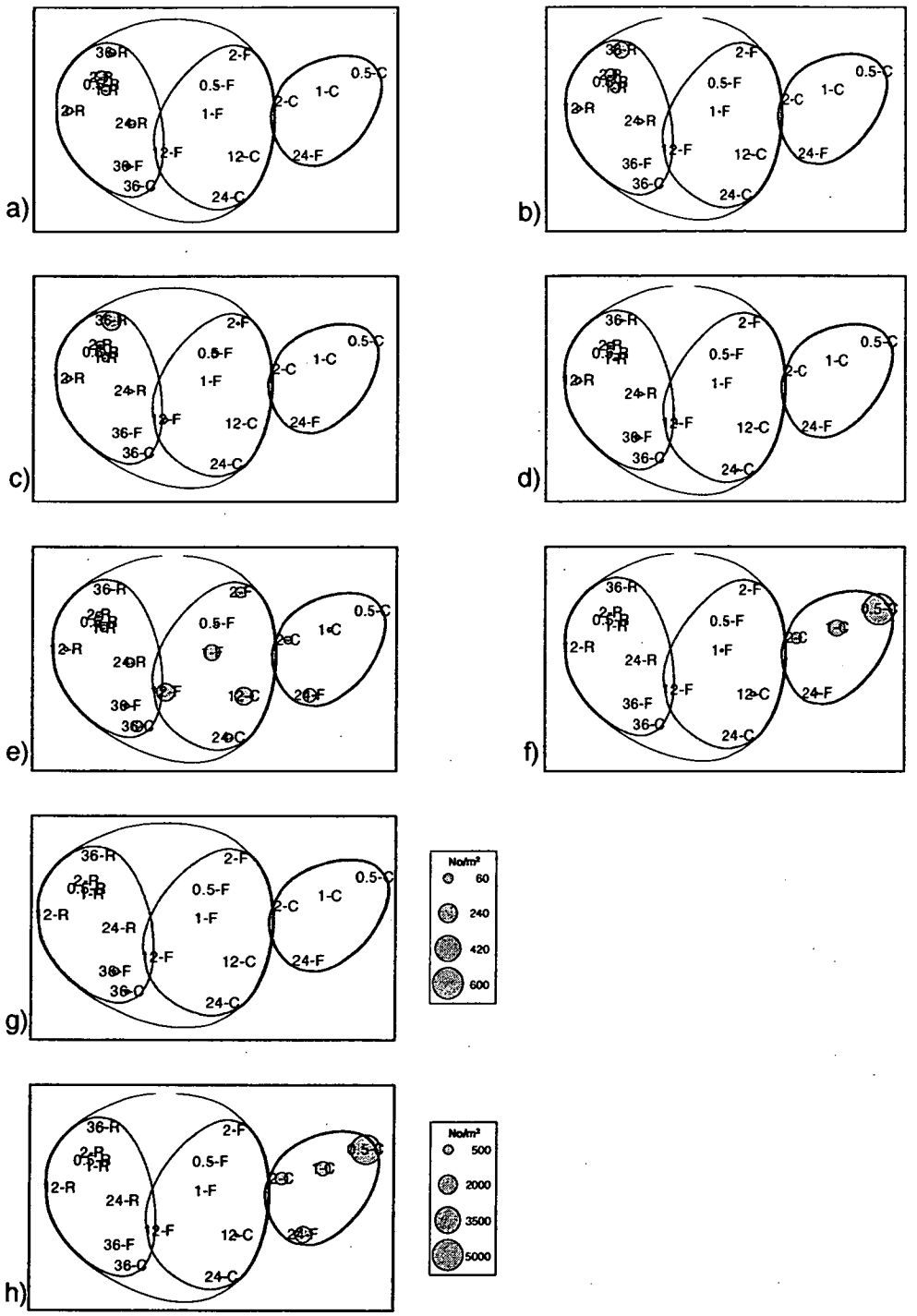


Fig. 3.2 Relative abundance of indicator species, a) *Lysilla jennacubinae, b) Nucula pusilla, c) Amphiura elandiformis, d) Lumbrineris* sp., e) *Euphilomedes* sp.1, f) *Malacoceros tripartitus,* g) *Paraprionospio coora* and h) *Capitella capitata* complex, at all stations and sample times. Scale on bubble plots a) - g) = 60, 240, 420 and 600 indivs/m^2, and on plot h) = 500, 2000, 3500 and 5000 indivs/m^2. Numbers indicate the month that samples were taken.

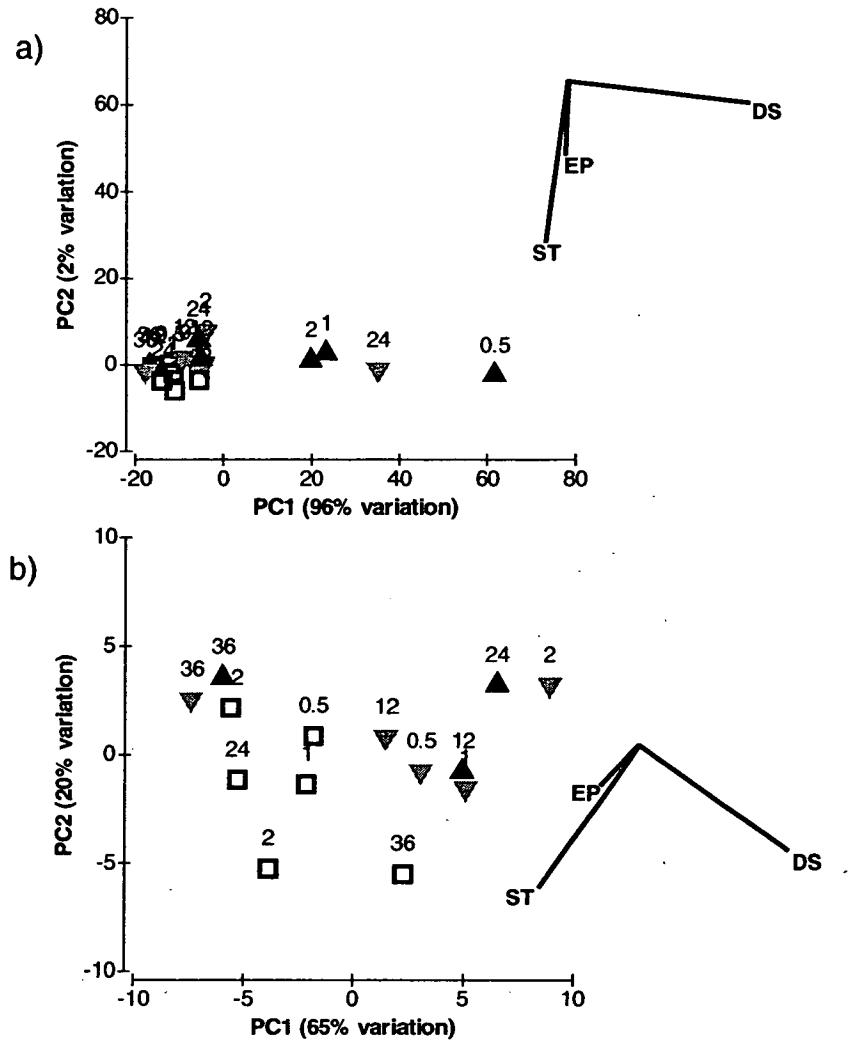


Fig. 3.3 Principal components analysis with biplot showing the sediment functions most strongly associated with site separations of reference (□), farm (▽) and cage (▲) stations for the dominant taxa over time using a) all sample stations and b) excluding the most impacted stations (four stations shown on right in panel a)). The proportion of the overall variability explained by each principal component is indicated on the relevant axis. Key sediment roles are indicated as: (sediment bioturbators (DS), consolidators/ stabilisers (ST) or epibenthic species (EP)). Numbers indicate the month that samples were taken.

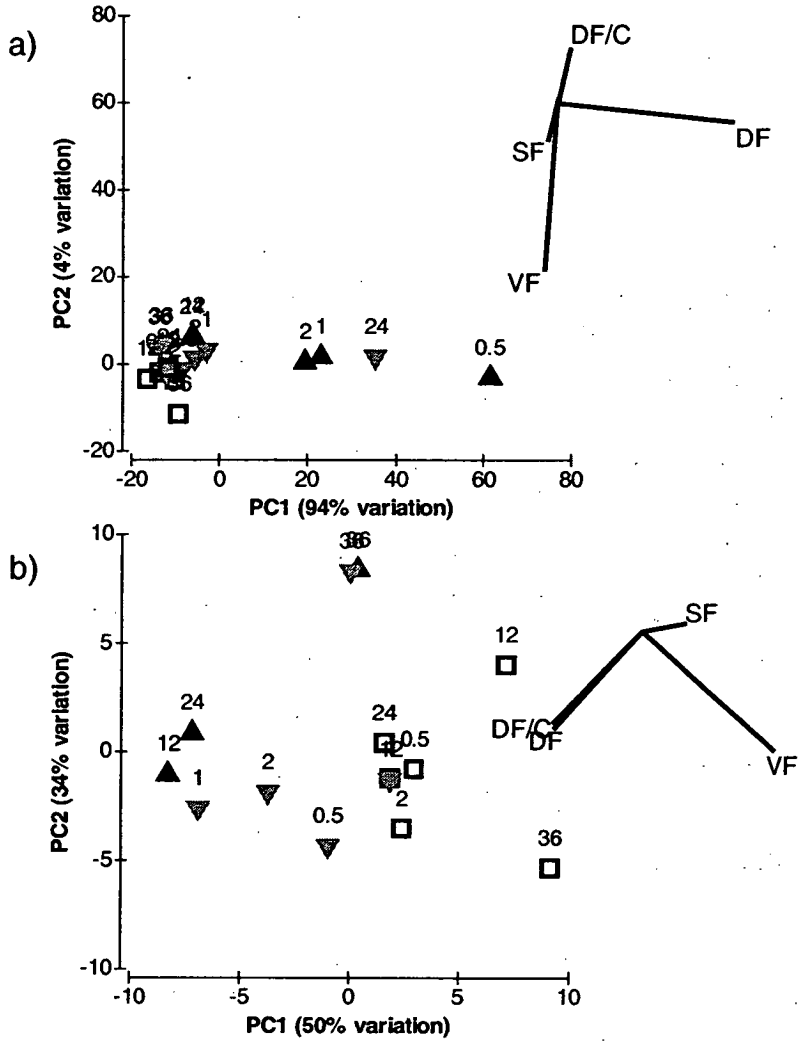


Fig. 3.4 Principal components analysis with biplot showing the feeding strategies most strongly associated with site separations of reference (□), farm (▽) and cage (▲) stations for the dominant taxa over time using a) all sample stations and b) excluding the most impacted stations (four stations shown on right in panel a)). The proportion of the overall variability explained by each principal component is indicated on the relevant axis. Key feeding strategies are indicated as: (predatory carnivores (C), suspension feeders (SF), deposit feeders, (DF) or taxa with a variable feeding strategy (VF)). Numbers indicate the month that samples were taken.

The greatest change in ecological function was associated with the cage communities in the initial recovery phase (i.e. cluster group A) (Fig. 3.3a, 3.4a, 3.5a). More than 95% of the total variability in the full dataset was associated with changes

between these initial communities and the remaining sites and times (principal component 1). These initial communities were functionally very different to both the references and to cage and farm stations sampled later in the study (Fig. 3.3a, 3.4a, 3.5a). The impacted communities were dominated (both numerically and in number of taxa) by deposit feeders, which would destabilize the upper sediments (Fig. 3.2). On the whole these species were opportunistic with the ability to rapidly reproduce and colonise, producing both benthic and pelagic larvae (Fig. 3.4a). After 12 months the function of the cage communities appeared to have largely returned to that of the reference communities (Fig. 3.3a, 3.4a, 3.5a).

Exclusion of the data for the most impacted stations (group A) from the analysis allows both the functional differences between the remaining communities and the subsequent recovery response to be examined more closely (Fig. 3.3b, 3.4b). The first principal component (PC1) accounts for almost 50% of the overall variability and associated with it, a strong gradient in functional response remains. The reference communities are positioned on the right of the plot whilst the earlier farm stations appear on the left. The sediment function of these stations separated along this impact gradient in much the same way as in the complete dataset. The farm stations (i.e. the more impacted end of the gradient) continued to contain a greater proportion of deposit feeding sediment destabilisers than the reference communities which had comparatively more suspension feeding and epibenthic sediment stabilizers (Fig. 3.3b). However, the gradient in reproductive function was slightly different. In numerical terms, the farm stations were most strongly associated with taxa where staged reproduction and production of pelagic larvae are the reproductive strategies (Fig. 3.4b). The communities at the cage station at 12 months were more strongly associated with opportunistic reproductive strategies. An increased prevalence and abundance of taxa with staged reproductive strategies resulted in the somewhat anomalous association of the reference station at 36 months with the early farm stations rather than the other reference samples.

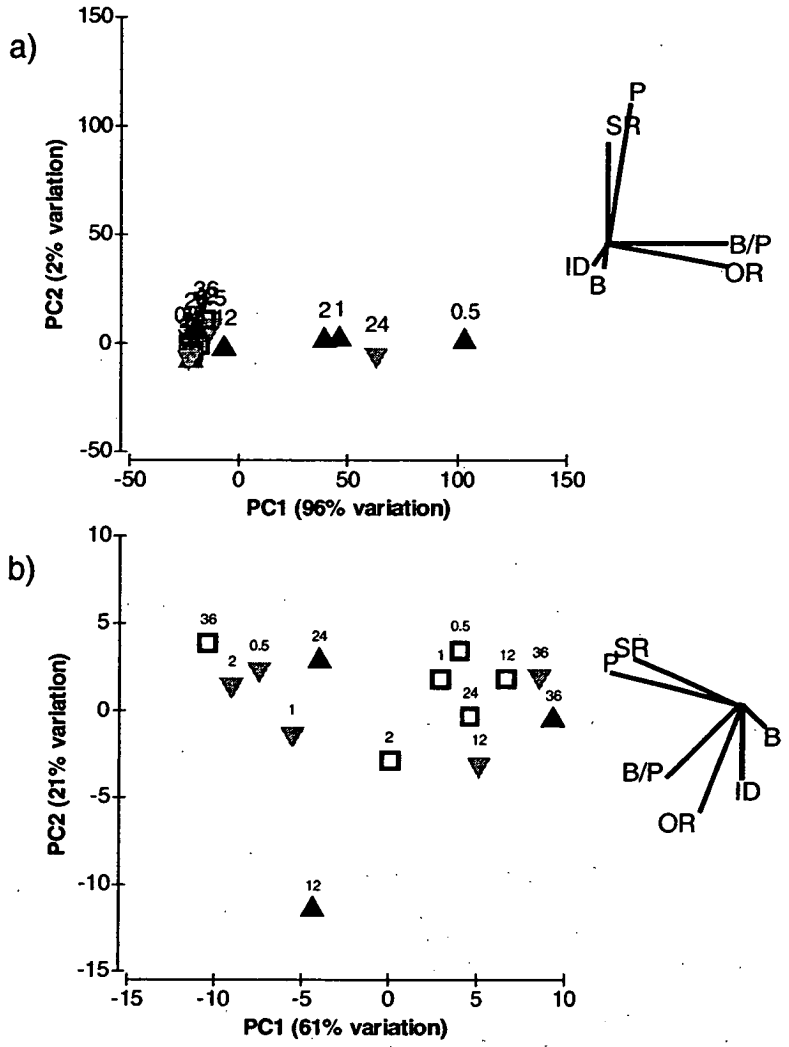


Fig. 3.5 Principal components analysis with biplot showing the reproductive strategies most strongly associated with site separations of reference (□), farm (▽) and cage (▲) stations for the dominant taxa over time using a) all sample stations and b) excluding the most impacted stations (four stations shown on right in panel a)). The proportion of the overall variability explained by each principal component is indicated on the relevant axis. Reproductive strategies indicated as: staged (SR), opportunistic (OR) or indeterminate (ID) reproductive cycles and benthic (B) or pelagic (P) larvae. Numbers indicate the month that samples were taken.

Table 3.4 The abundance, ecological function and reproductive strategies of the ten species which contributed most to the overall group similarity of the reference and cage communities. Ecological function has been derived from literature sources; where no species specific information available the nearest taxonomically similar species/group was used, these results are indicated in italics.
Key to Ecological Function: LG-Large bodied animal (likely to be retained on a 10mm sieve), MD-Medium sized animal (retained on a 4mm sieve), SM- small animal (retained on a 1mm sieve), SF- Suspension feeder, DF-Deposit Feeder, C- Carnivorous, S-Sediment stabiliser, D-Sediment Destabiliser, E-Epibenthic.
Key to Reproductive Strategy: SG – Single staged generation, MG – Multiple generations, P-Planktonic juveniles, B-Benthic larvae, SR-Staged reproductive cycle, OR- Opportunistic reproductive cycle.

Reference (Average group similarity 58.46)

Species ID	Phylum	Simil Contrib%	No/m^2	Ecological Function	Reproductive Strategy
Nucula pusilla	Mollusca	14.86	61	*LG: DF: D*	*MG:P:SR*
Lysilla jennacubinae	Annelida	14.52	46	*LG: DF: S*	*Unknown*
Amphiura elandiformis	Echinodermata	12.68	53	*LG: DF/SF: D*	*MG:P:SR*
Euphilomedes sp.1	Crustacea	9.34	19	*SM: DF: D*	*Unknown*
Lumbrinereis sp.1	Annelida	7.71	13	*LG: DF:D*	*SG;B;SR*
Nassarius nigellus	Mollusca	7.09	14	*MD: DF: E*	*MG:B:SR*
Corbula gibba	Mollusca	6.50	10	MD: DF: D	MG:P:SR
Theora fragilis	Mollusca	6.22	20	*MD: DF: D*	*MG:P:SR*
Euphilomedes sp.2	Crustacea	6.16	8	*SM: DF: D*	*Unknown*
Nemertea sp	Nemertea	3.29	6	*MD: C/DF: D*	*MG:B:SR*
Euchone limnicola	Annelida	3.22	4	*SM:SF:S/D*	

Impacted (Average group similarity 60.52)

Species ID	Phylum	Simil Contrib%	No/m^2	Ecological Function	Reproductive Strategy
Capitella capitata (complex)	Annelida	53.17	1,700	SM: DF: D	MG,B/P,OR
Malacoceros tripartitus	Annelida	10.17	140	*MD: DF: D*	*MG: Unknown: OR*
Theora fragilis	Mollusca	5.96	29	*MD: DF: D*	*MG:P:SR*
Prionospio kulin	Annelida	5.56	25	*MD: DF: D*	*SG:P:Unknown*
Euphilomedes sp.1	Crustacea	5.40	31	*SM: DF: D*	*Unknown*
Corbula gibba	Mollusca	4.36	18	MD: DF: D	MG:P:SR
Nassarius nigellus	Mollusca	4.02	18	*MD: DF: E*	*MG:B:SR*
Simplisetia amphidonta	Annelida	4.02	11	*MD: DF/C: D*	*SG:P:SR*

3.4 Discussion

In the first few months post-farming the cage conditions were highly impacted, falling into the "polluted" category (Pearson and Rosenberg, 1978). The community was dominated by two species, *Capitella capitata* and *Malacoceros tripartitus;* both of which are opportunists able to tolerate extremely high levels of organic carbon and increased sedimentation rates (eg Johannessen *et al.*, 1994; Henderson and Ross, 1995, Levin, 2000). Although *C. capitata* and *M. tripartitus* defined the cage communities over the first two months, after 12 months both species were all but absent from the cage communities, indicating a marked improvement in environmental conditions. Overall, there was a gradient of recovery in the benthic infaunal community structure over the fallow period consistent with established models (Pearson and Rosenberg, 1978, Rhoads *et al.*, 1978).

As the sediment conditions improved the abundance of the impact indicators declined and overall diversity increased. The transitional communities represented a broad continuum of change with complex species interactions, and as such it was difficult to identify species representative of the whole community. However, there were some species whose abundance increased during the initial recovery period relative to the impacted community. The native dog whelk, *Nassarius nigellus,* a common epibenthic scavenger (Edgar, 1997), was amongst the first species to return to the cage sediments, taking advantage of the increased food supply. However, abundances declined again towards the end of the study, presumably as resources became depleted. The introduced bivalve, *Theora lubrica,* is a selective deposit-feeder which is also relatively tolerant of organic pollution, hypoxic conditions and high levels of sedimentation (Tamai, 1996, Saito *et al.*, 1998, Talman, 1998). It too colonized the impacted sites very quickly after farming stopped, and was more common in recovering sediments than in either the impacted or reference communities, but as with *N. nigellus* numbers declined as sediment conditions returned to normal. Consequently, the presence of these species in increasing abundance after cessation of impact may be indicative of the transitional community and the onset of recovery.

Throughout the recovery period the infaunal community structure at the cage

and farm stations continued to change, becoming more like the reference conditions but even after 36 months fallowing the communities did not fully recover to reference conditions (Chapter 2). Other studies of recovery after fish farming have also suggested that complete recovery (i.e. return to reference community structure) can take many months (Lumb, 1989, Ritz *et al.*, 1989, Johannessen *et al.*, 1994, Pohle *et al.*, 2001, Pereira *et al.*, 2004) or even years (Karakassis *et al.*, 1999, Brooks *et al.*, 2004). This is perhaps not surprising as complete recovery requires the full re-establishment of the unimpacted community. How quickly this occurs will be determined to a large extent by the nature and resilience of the natural background community (Bonsdorff, 1989) and will take as long as the life cycle of the dominants in that background community (McCall and Tevesz, 1983). In the present study the climax communities contained several species from taxa known to have relatively long life expectancies and which can take several years to mature. Brittlestars of the genus *Amphiura* are often long lived; *A. chiajei* has a lifespan in excess of 10 years (Munday and Keegan, 1992) whilst *A. filiformis* is known to live more than 20 years (O'Connor *et al.*, 1983, Sköld *et al.* 2001) with neither species becoming reproductively mature until they are 3-4 years old (Sköld *et al.* 2001, Fish and Fish, 1996). Similarly species of *Nucula* are known to have a lifespan of more than 10 years (Wilson, 1992) and do not mature until their second year (Davis and Wilson, 1983).

Unimpacted "normal" conditions had a diverse fauna containing many species sensitive to the adverse effects of organic enrichment, i.e. reduction in sediment oxygen levels, increases in sediment organic content and sedimentation rate. Four of the dominant species in this community; the brittle star, *Amphiura elandiformis,* two polychaetes, *Lysilla jennacubinae* and *Lumbrineris* sp.1, and a bivalve *Nucula pusilla* would be highly susceptible to the direct effects of organic enrichment, either as a result of their low tolerance to oxygen or sedimentation (Hutchings, 2000, Paxton, 2000, Crawford *et al.*, 2002). The reproductive ecology of these species would also affect their ability to re-establish after impact. *Amphiura* generally have pelagic larvae and a staged reproductive cycle (Bowmer, 1982, Pedrotti, 1993, Hendler and Tran, 2000, Tominaga *et al.*, 2004). Similarly, terrebelids tend to have a relatively restricted spawning season of only a few days or weeks (Hutchings, 2000), whilst Lumbrinerids

tend to be fairly long lived with low reproductive effort (Fauchald, 1983). As a consequence they all would be slow to recover where their populations had been compromised.

In the present study although the community structure at the cage and farm stations never returned fully to that of the references (Chapter 2), many reference species returned soon after active enrichment ceased. However, the environmental sensitivities of adult and larval stages can be very different, with larval stages often being less tolerant (Snelgrove and Butman, 1994) than their adult counterparts. Migration of juvenile and adult life stages rather than larval settlement is the most common early recolonisation process and is particularly important for species possessing late successional stage life history traits (Whitlach *et al.*, 2001). This may explain the initial increase in the numbers of reference species at the cage and farm stations. Larval recolonisation often encompasses a broader range of species than would be available from direct immigration, and some of these species may be better adapted to the prevailing environmental conditions than the existing fauna (Thrush and Whitlach, 2001). This might explain the decline in the number of reference species and increase in other taxa at the cage and farm stations in second and third years. In studies where recovery has been shown to be particularly rapid, occurring within only a few weeks or months, (e.g. Ritz *et al*, 1986, Lu and Wu, 1998) it may be that only the initial migration of fauna has occurred. This raises some interesting questions regarding how recovery is evaluated and at what point recovery is determined to have occurred. Where there is a possibility of regression or a divergence in the recovery trajectory, studies of recovery need to be undertaken over a sufficient temporal scale to determine whether community changes are sustained. Under these circumstances ecological function may be a more robust measure of recovery.

Absence of the rarer climax species may be important in terms of the overall biodiversity of the system, but may not be as important in defining the overall function of the system. The main ecological functions of the background community were re-established at the cage stations after only 12 months. Although the length of time required for full recovery may vary, once the recovery process is established the system will progress to an equilibrium state providing no further perturbation is encountered

(Young *et al.*, 2001). Consequently, functional recovery of the system may be a more useful reference point for establishing and managing recovery than restoration of community composition, as functional recovery suggests that the balance of the ecological processes within the system has returned. At 12 months the cage stations were functionally similar to reference stations, in terms of the sediment role, feeding and reproductive strategies of the communities. However, at this point only 53% of the reference species had re-established at the cage stations, which suggests that functionally there was a large amount of redundancy in the species information. Although the taxa associated with each successional stage in the enrichment models will vary with geographic location, the organism-sediment relationships and changes in trophic structure appear to be similar (Pearson and Rosenberg, 1978, Rhoads and Germano, 1982). Consequently where broader spatial comparisons are of importance then the specific functional types associated with each stage are the units of interest rather than the individual species (Rhoads and Germano, 1986). The primary difficulty with using functional type is whether there is enough information on the biology and ecology of the local fauna to successfully establish the functional status of the system. However, in the current study the fauna was dominated by the 10 most abundant species; these species were largely the local dominants and as such their ecology and biology, or at least that of taxonomically similar species, was quite well known.

The functional response to recovery followed a similar gradient to the community changes, and as with community succession the extremes were readily distinguishable, i.e. impacted and unimpacted conditions, but the transitional communities were less well defined. Comparing the functional and community gradients enables characterization of the functional criteria associated with the successional changes. Impacted conditions were evident where more than 70% of the dominant fauna (i.e. those species that made up more than 70% of the overall community similarity) displayed opportunistic strategies. Under unimpacted conditions the level of opportunists in the dominant fauna fell below 10%, stabilizing species comprised more than 20% of the dominant fauna and feeding strategies other than deposit feeding accounted for more than 25% of the fauna. We were unable to identify any studies specifically comparing impact or recovery level in any quantitative or semi-

quantitative way with the functional structure of the community. Consequently, it was not possible to determine the relevance of these criteria in a broader spatial context, but it would be useful to see if the criteria hold up in other areas.

Comparing responses across systems, particularly those with different hydrodynamic conditions, is difficult (Thrush and Whitlach, 2001). Differences in the background ecology may result in major differences in the successional endpoints (Rothschild *et al.*, 1994), accordingly, the functional criteria defining recovery may vary between locations. However, recent studies comparing regional differences in impact levels and benthic community structure in Tasmania suggested that there were regional parallels in community response within current farming areas (Edgar *et al.*, 2005) and that there were similar community types between regions (Macleod and Helidoniotis, 2005). This suggests that where the background community types are known then it should be possible to characterize functional response criteria. This process would be greatly facilitated by further information on the basic biology and ecology of the local fauna. The Australian benthic fauna is on the whole very poorly known and further information on the ecological function of even the most common species would be extremely valuable.

The findings of this study identify some important points for considerations by environmental managers. The criteria used to define recovery must be clear and relevant to the environmental management goals. Differences in the successional endpoints of natural communities will markedly affect the timeframe over which recovery should be assessed, consequently the ecology of the unimpacted environment must be properly understood. Finally, in relation to the recovery response, functional recovery may represent a critical reference point. Where there are to be no further impacts on the environment, functional recovery may be considered to represent the point at which a system has demonstrated its capacity to recover.

Chapter 4:

Evaluation of Short-Term Fallowing as a Strategy for the Management of Recurring Organic Enrichment Under Salmon Cages.

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Macleod, C.K., Moltschaniwskyj, N.A., and Crawford, C.M. Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages.

Abstract

Rotation of cages within fish farm leases and the subsequent fallowing of areas of seabed is commonly used to allow recovery of infaunal communities following periods of organic enrichment. To investigate the effect of different background environmental conditions on recovery response, two Atlantic salmon (*Salmo salar* L.) fish farm sites in southeast Tasmania were sampled over two commercial fallowing cycles. Despite similar stocking levels and feed input there were significant differences in the way in which sediment at each farm responded to the cessation of fish stocking. Sediments at both farms showed some improvement in the community structure over a three month fallow period, but the community structure only recovered to that present before stocking not to that at the reference sites. The similarity of the impact sites to the reference sites increased from ca. 25% to 31% at one site and 11% to 27% at the other after fallowing. Rate and extent of recovery were affected by farm location, initial impact of the sediments, and length of fallow period. Initial recovery was faster at the more sheltered site than at the more exposed site, possibly reflecting differences in environmental resilience with the more sheltered location better able to assimilate organic inputs. Accordingly general fallowing management protocols may need to be adapted to reflect differences between sites. The findings of this study suggest that the recovery response of benthic communities can be predicted once baseline conditions are understood.

Keywords: Tasmania, organic enrichment, recovery, aquaculture, benthic infauna, fallowing

4.1 Introduction

Faecal waste and uneaten feed from commercial finfish cage aquaculture results in organic enrichment of the underlying sediments (Black, 2001). To alleviate impacts on the sediments and to give the sediment an opportunity to recover, fish-holding cages are often removed or are left fallow for a period of time. Although environmental monitoring of the sediments is mandatory, there is no legislative requirement for fallowing in Tasmania. Environmental regulations only require that there be no "unacceptable impact" associated with farming practices (Woods *et al.*, 2004). Therefore the duration of any fallow period is largely at the discretion of the farm manager. The amount of time actually required for sediment remediation is poorly understood at present. Three months of fallowing is generally considered to be a reasonable timeframe and is regularly used.

Many studies have examined the temporal and spatial effects of organic enrichment from cage aquaculture on the benthic community structure. It takes a relatively long time for the biota to recover fully (Karakassis *et al.*, 1999, Pohle *et al.*, 2001, Brooks *et al.*, 2004, Pereira *et al.*, 2004). Nonetheless, it is evident that, in all but the very worst cases, recovery of the sediments commences fairly quickly once farming has ceased (i.e. within a number of weeks). In one of the few studies undertaken in Australia on sediment remediation after cage fish farming, the benthic ecology of sediments at a relatively exposed Tasmanian site recovered to reference conditions after only seven weeks (Ritz *et al.* 1989). However, in contrast, and consistent with the majority of the literature, the benthic faunal community structure under cages at a relatively sheltered location (North West Bay, Tasmania) continued to differ from reference conditions 36 months after the cessation of farming (Chapter 2). The difference between the rates of recovery observed in the two Tasmanian studies was attributed partly to differences in background environmental conditions and partly to the expansion and intensification of the salmon farming industry since the initial study (Chapter 2).

In both Tasmanian studies, and in the majority of prior investigations, recovery of the sediments was gauged against a return to reference conditions. However, for fish

farming operations to be sustainable it is not necessary for sediment condition to return to a reference state. Recovery to the extent that it does not result in progressive chemical or biological deterioration of sediments may be sufficient to support long-term farming operations. In this regard there have been no studies that have investigated appropriate fallowing regimes for such practical management of environmental condition.

Most studies of sediment recovery associated with fish farming have been restricted to single sites/leases within similar geographic areas (Ritz *et al.*, 1989, Karakassis *et al.*, 1999, Pereira *et al.*, 2004), and there are very few aquaculture-based studies that specifically compare large-scale spatial variability in the sediment recovery response. One exception compared several farm sites and found significant differences in their biological recovery rate (Brooks *et al.*, 2003, 2004). Recovery at the site in the later study (Brooks *et al.*, 2004) was markedly slower, with the suggestion that it could take more than six years for biological recovery. Although the authors did not specifically examine why this was the case, they suggested that it may be linked to environmental differences.

Local environmental conditions can have a major influence on the rate at which sediments recover from organic enrichment (Black, 2001). Both physical and biological conditions will affect the rate of recovery response (Boesch and Rosenberg, 1981, Beveridge, 1987). Communities of highly stressed and physically variable environments may be less complex, but can recover more quickly from a disturbance than those of more benign and less variable areas (Bolam and Rees, 2003). Several studies have shown a direct relationship between the chemical condition of the sediment and the biological response (eg Holmer and Christensen, 1992, Hargrave *et al.*, 1997, Wildish *et al.*, 2001). Measurement of redox potential and sediment sulfide concentration have been recommended as potentially useful, cost-effective approaches for assessing sediment degradation (Hargrave *et al.*, 1997, Wildish *et al.*, 2001, Crawford *et al.*, 2002, Edgar *et al.*, 2005). However, the value of these approaches in relation to sediment recovery has not yet been clearly established.

The aims of this study were to assess the extent to which sediments recover

under the normal production protocols employed on a commercial salmon farm in Tasmania, and to determine the variation in recovery rates between sites with differing environmental characteristics. This was done by evaluating the environmental and farm management factors which may influence recovery response. In addition, the value of established chemical condition measures of degradation (redox potential and sediment sulfide concentration) was examined to ascertain the applicability of these measures to the assessment of sediment recovery.

4.2 Methods

4.2.1 Selection of sampling locations

Two farm locations in south eastern Tasmania, Creeses Mistake (Tasman Peninsula) and Stringers Cove (Port Esperance) were sampled in this study (Fig. 4.1). These farms are broadly representative of the differing environments in which Atlantic salmon culture is undertaken in southwest Tasmania. Creeses Mistake is a relatively exposed, shallow (20 m) and fully marine site whereas Stringers Cove is in deeper (40 m) more sheltered waters that are occasionally subject to the freshwater influence of the nearby Esperance River.

Sediment recovery associated with standard farm production protocols was studied over two annual production cycles. Both farms employed an annual stocking regime where cages were stocked for nine months and then fallowed for three months. At each farm the study cages were circular with a circumference of 120m. Sediment samples were collected from cage stations and references prior to the cages being stocked (TX), at the end of nine months of stocking (i.e. at the end of the stocked phase/start of fallow period - T0), and at the end of a three month fallow period (T3). In addition, during the second year samples were collected from both cage and reference stations at monthly intervals during the fallow period.

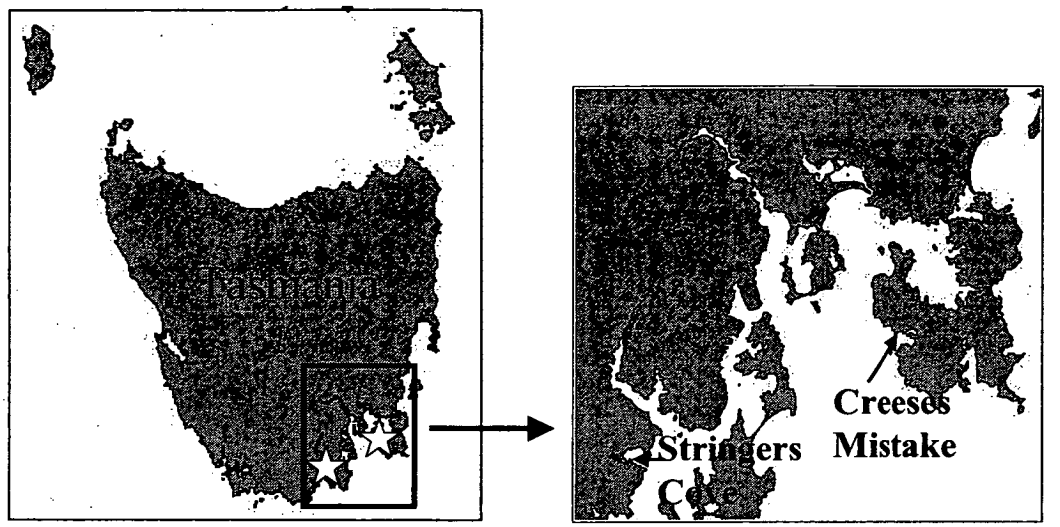

Fig. 4.1 Location of study farms in south eastern Tasmania. Creeses Mistake is located in Wedge Bay on the Tasman Peninsula and Stringers Cove is within Port Esperance.

During the second annual production cycle, cages were restocked at Creeses Mistake in exactly the same position as in the previous cycle (stations 5 and 8, Fig. 4.2) however at Stringers Cove the cages were restocked adjacent to the stations used in the first cycle (i.e. cages were at stations 1 and 2 in first production cycle and at stations 1A and 2A in second cycle, Fig. 4.2). The Creeses Mistake cage stations had previously been farmed whereas cage stations 1, 2, 1A and 2A at Stringers Cove had not. Consequently, previously farmed cage stations 3A and 4A at Stringers Cove (Fig. 4.2) were analysed as replicates in the second annual farming cycle but not the first annual cycle.

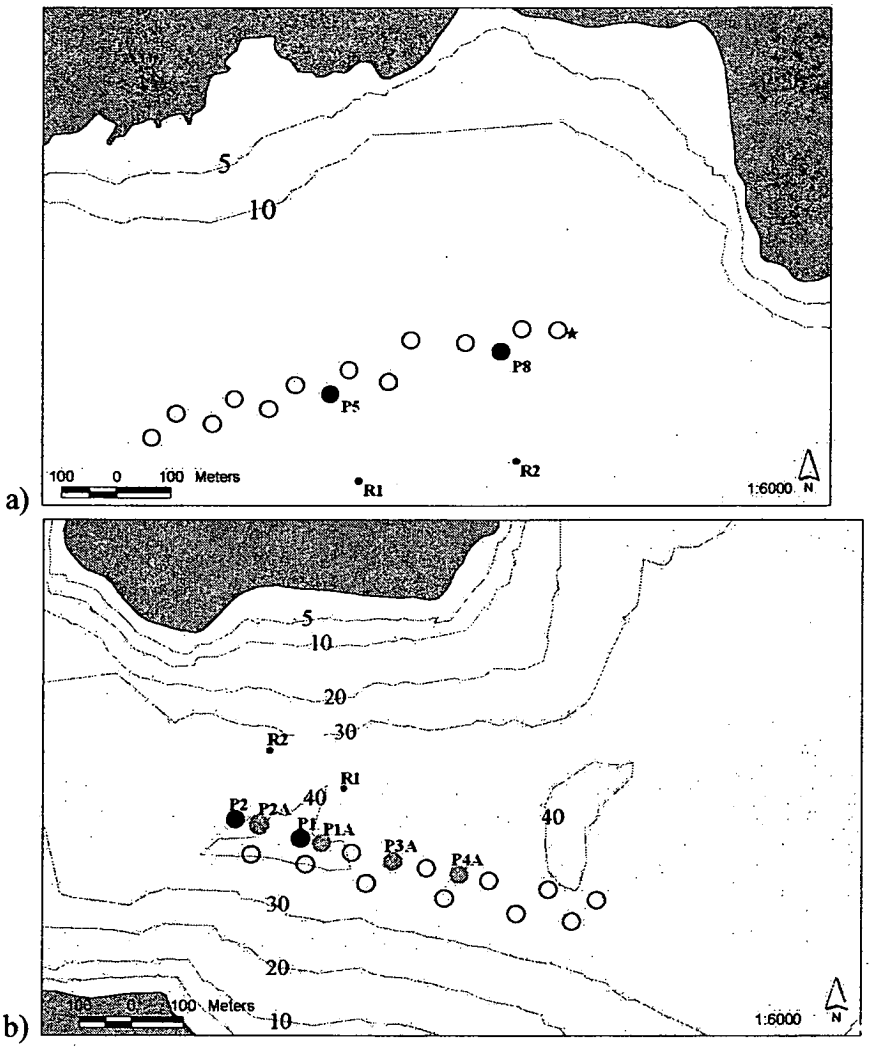

Fig. 4.2 Cage stations and reference sites for sediment samples for a) Creeses Mistake and b) Stringers Cove. Stringers Cove sample sites shown as ● were stocked in the 1st production cycle whilst those shown as ◎ were stocked in the 2nd cycle. Sites shown as ○ indicate the positions of other cages in the first production cycle.

Prior to sampling, each farm area was mapped using a Garmin 135 GPS Map unit coupled with a Racal differential unit. Depth and positional information were collected for all cages present on the lease at the time. In addition, reference locations, within the same depth range, but 150 m distant from the edge of selected study cages cages and cross-current from the lease were located using the depth contours and GPS.

4.2.2 *Faunal sampling:*

Five replicate samples were collected from each cage station and reference using a Van Veen Grab (surface area – 0.0675 m^2) which collected between 2-4 litres of sediment and sampled to a depth of 15-20 cm. Grab contents were transferred to mesh bags (mesh size 0.875 mm) and rinsed. Samples were then wet seived to 1 mm and the retained material preserved in a solution of 10% formalin:seawater (4% formaldehyde). Samples were transferred to the laboratory for sorting and the infauna identified to the lowest possible taxonomic level and enumerated.

4.2.3 *Redox & Sulfide Assessment:*

Three replicate cores (perspex tubes 250 mm length x 45 mm internal diameter) were taken using a Craib corer at each cage station and reference site for measurement of redox potential and sulfide concentration. Redox and sulfide were measured at 3 cm depth using a WTW Redox Probe and a Cole-Parmer 27502-40 silver/sulfide electrode respectively. Sulfide was sampled according to the method described by Wildish *et al.* (1999), with 2 ml of anti-oxidant buffer added to a 2 ml sediment sample prior to measurement.

4.2.4 *Statistical analysis:*

As the aim of the present study was to evaluate the rate and extent of recovery associated with farm fallowing three levels of recovery were considered:

Level 1 recovery - improvement in sediment condition (i.e. biologically and chemically)
Level 2 recovery - return to pre-stocking sediment condition (i.e. recovered conditions within an operational farm)
Level 3 recovery - return to reference sediment condition

Abundance data were square root transformed to reduce the influence of abundant taxa and the Bray-Curtis similarity index was used because of the robustness of this statistic to zero-inflated data sets (Clarke, 1993). Replicate samples were used to generate a mean value for each cage and reference site.

An ordination plot using non-metric multidimensional scaling (nMDS) was used to identify differences in the community structure between reference sites and cages at each farm over the fallow period. The significance of differences between conditions prior to stocking, at the reference stations, and at the end of the fallow period was assessed using the ANOSIM randomisation test provided by the Plymouth Routines in Multivariate Ecological Research (PRIMER) software package (PRIMER, 2006).

Initial impact levels were assessed using the Bray-Curtis similarity of the full species abundance (square root transformed) data set between cage stations and associated references immediately prior to fallowing, i.e. cages that had a community structure with a high similarity to their respective reference site were considered to be less impacted that cages that had a low similarity to their respective reference site. The Bray-Curtis similarity matrix was also used to determine the relationships between farm location, farming practices and the period of time that the site had been fallowed. This analysis was used to examine absolute changes over time and the full species dataset was used to allow detection of subtle compositional changes in community structure during the relatively short fallow period. The Bray-Curtis similarities between the start (T0) and end (T3) of fallowing, between the end of fallowing (T3) and pre-stocking (TX), and between recovered condition (T3) and reference condition at the end of fallow period (R3) were used as relative measures of level 1, 2 and 3 recovery respectively.

The rate of change in the community structure at each cage during the fallow period was determined from the gradient of the regression line generated from changes in the monthly Bray-Curtis similarity over the fallowing period. Univariate statistical analysis of similarity measures was used to examine differences in recovery over time. Student t-tests were used to compare the average rate and magnitude of change in the community structure between the farm locations. Regression analysis was used to examine the rate of change at the references and cages within farms over the fallow period as a function of initial impact. The association between initial impact, farming factors (feed input and number of adjacent cages), and rate and level of change were examined using Pearson correlation coefficients.

Comparisons of the average redox potential and sulfide concentration between cage and reference sites (across both farms) were made using Student t-tests.

4.3 Results

At both farm locations the community structure of the cage stations changed during the fallow period in a manner that reflected recovery from organic enrichment. Over the three month fallow period (i.e. between T0 and T3) the community structure at all cage stations changed significantly (ANOSIM, Rho = 0.25, p = 0.016), with cage stations at both farms recovering to pre-stocking condition (ANOSIM, Stringers – Rho = 0.100, p=0.185, Creeses – Rho = 0.41, p = 0.114) (Fig. 4.3). However, neither farm location recovered to reference conditions (ANOSIM, Stringers – Rho = 0.92, p <0.00, Creeses – Rho = 0.80, p <0.001) (Fig. 4.3).

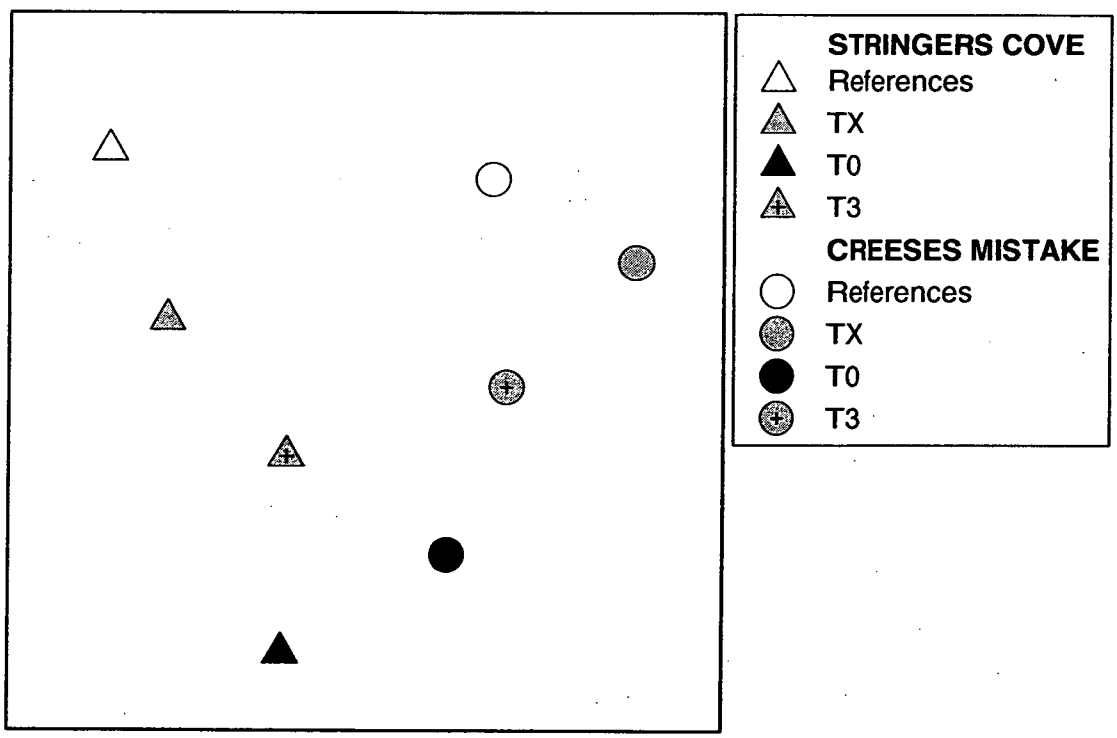

Fig. 4.3 Non-metric multidimensional scaling plot of community similarities at the two farms. Stress = 0.03. (TX – pre-stocking, T0 – start of fallow period, T3 – end of 3 month fallow period).

The reference communities were very different between farm locations (ANOSIM, Rho = 1.000, p <0.001), but within farms the reference communities were similar both spatially and temporally (Table 4.1). Regression analysis of the monthly Bray-Curtis similarities for the reference stations over the fallowing period shows no evidence of any significant variation in the reference community structure at either farm location over the fallow period (Creeses Mistake: $r^2 = 0.64$, F = 1.81, df 1,2, p = 0.407; Stingers Cove: $r^2 = 0.78$, F = 3.59, df 1,2, p = 0.309). At Stringers Cove the changes in the cage communities over the fallow period (T0-T3) were greater than in the reference communities (t = –4.56, df 6, p = 0.004), i.e. similarity levels were significantly lower. At Creeses Mistake the overall change at the cage sites was less conspicuous and was within the range of the reference communities over the same period (t = –1.19, df 4, p=0.301) (Table 4.1).

Table 4.1 Average similarity (± s.e.)(using Bray-Curtis similarity index) a) between cage stations and equivalent references at the start of the fallow period (T0-R0), b) between the cage stations at the start and end of the fallow period (T0-T3), c) rate of change as the gradient of change in similarity determined from the change in similarity between the cages each month over the fallow period. d) average daily feed input (± s.e.) and e) average number of adjacent stocked cages (± s.e.) over period that the cage was stocked. Monthly data was unavailable from Creeses Mistake or references in first production cycle. (Note larger numbers indicate greater similarity levels and therefore less change).

Station	Similarity between T0 & R0	Similarity between T0 & T3	Rate of change*	Average Feed Input (kg/day)	Average No of Adjacent Cages
Stringers					
PC1-Cages	8.50(±1.24)	15.45(±7.05)	11.85(±2.78)	519(±20.0)	1.41(±0.39)
PC2-Cages	13.74(±2.60)	29.80(±6.00)	5.84(±5.05)	324(±11.3)	1.49(±0.24)
PC1-Refs		63.17(±1.00)			
PC2-Refs		68.24(±0.01)	0.54(±0.97)		
Creeses					
PC1-Cages	18.31(±3.33)	49.63(±4.78)		491(±36.0)	1.75(±0.09)
PC2-Cages	31.29(±3.35)	31.16(±3.07)	-5.75(±6.01)	304(±52.5)	1.69(±0.03)
PC1-Refs		59.03(±0.35)			
PC2-Refs		50.77(±0.89)	2.62(±8.46)		

Overall the two farms were similar in the degree to which they differed from the references at the end of the fallow period (T3-R3) ($t = 0.65$, $df = 3.5$, $p = 0.554$) and in the extent of the change in the community structure that occurred over the fallow period (T0-T3) ($t = 1.93$, $df = 8$, $p = 0.089$) (Fig. 4.4). However, the average rate of change differed between the two sites ($t = -2.33$, $df = 7.9$, $p = 0.048$). Whilst at both locations the community structure showed a level of recovery, at Stringers Cove the rate of change over the 3 month fallow period was positive whilst at Creeses Mistake it was negative. A positive rate indicates that the community became more similar with each subsequent month, whereas a negative rate indicates that the similarity levels decreased in each subsequent month. For this to occur, and still be indicative of recovery, it is clear that there must be differences in the components of the community affected, i.e. a positive response suggests that the species representing unimpacted conditions are increasing whilst a negative rate suggests that the species indicative of the impacted community are decreasing. The greater the magnitude of the rate the faster the recovery in the community structure occurred.

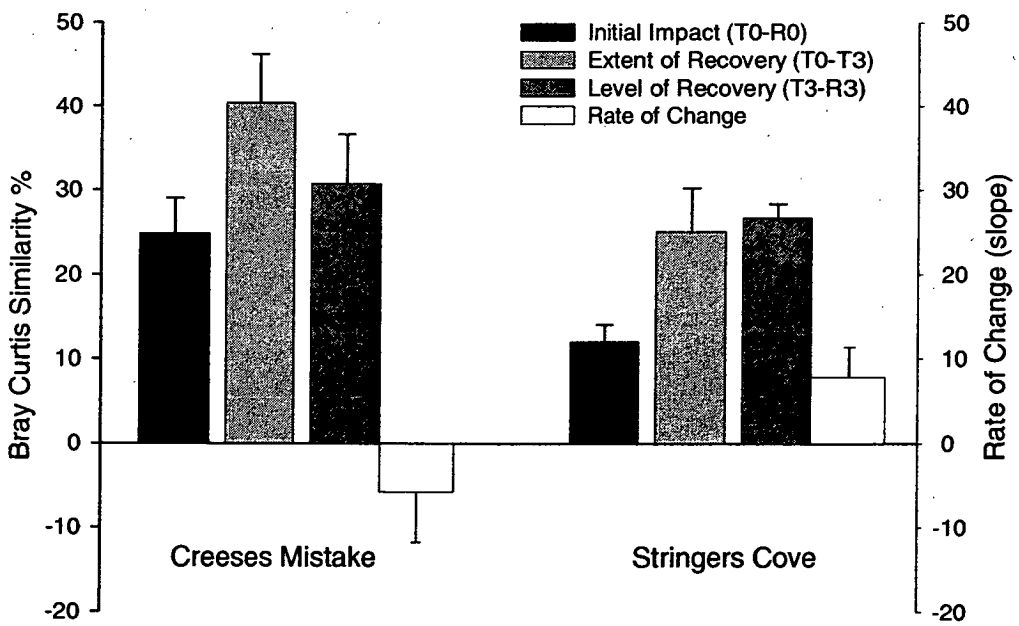

Fig. 4.4 Mean Bray-Curtis similarity (± SE) for initial impact (T0-R0), extent of impact (T0-T3), level of recovery (T3-R3), and the mean rate of change (± SE) at Creeses Mistake and Stringers Cove.

The community structures and recovery response differed markedly between farm locations (Fig. 4.3). The community structure prior to recovery was most similar between farms when the impact levels were greatest (i.e. T0), but even at this stage there were still significant differences between the farms (ANOSIM, Rho = 0.83, p = 0.006), and these differences persisted over the fallow period (ANOSIM, Rho = 0.75, $p<0.001$). Initial impact communities at each of the farm stations were markedly different (ANOSIM, Rho = 0.47, p = 0.003). However, the farms were similar in the extent to which the initial impact communities differed from their respective reference communities (T0-R0) (t = 2.74, df 1,7, p = 0.05) (Fig. 4.4). Initial impact affected the rate of change during fallowing ($r^2 = 0.58$, F = 8.25, df 1,7, p = 0.028). When the initial impact was greater (i.e. Bray-Curtis similarity between T0 and R0 was small) the rate of change over the fallow period was faster (Fig. 4.5).

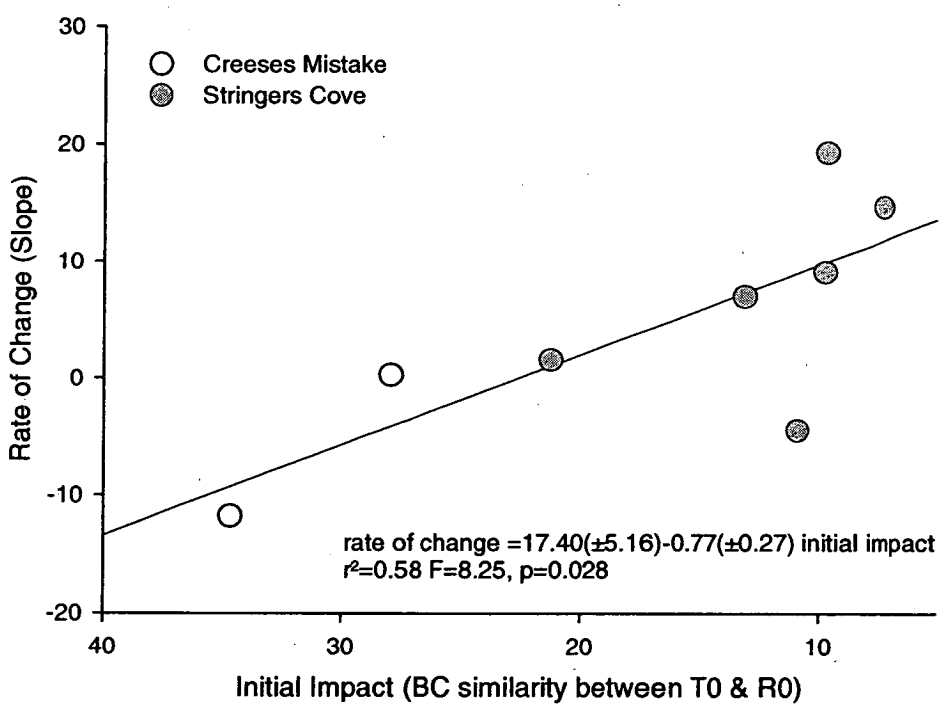

Fig. 4.5 Initial impact versus rate of change over fallow period (slope of regression for monthly similarity levels). Note that low initial impact levels correspond to high similarity levels (T0-R0) whilst high initial impacts are consistent with low similarity levels (T0-R0). Values in brackets are SE's of regression parameters.

The magnitude of the initial impact also affected the magnitude of the change in the community structure over the fallow period (i.e. Bray-Curtis similarity between T0 and T3) (Fig. 4.6). In this case the data suggest that there may be difference in the response at Creeses Mistake from that at Stringers Cove. There was a strong linear relationship between initial impact and extent of recovery at Stringers Cove ($r^2 = 0.84$, $F = 27.33$, df 1,5, $p = 0.006$). When the initial impact was greatest the extent of change in community structure over the fallow period was also greater. The cage stations which had the highest initial impact (i.e. differed most from the references prior to fallowing (T0-R0)) recovered most over the fallow period (i.e. had the lowest similarities (T0-T3)), whilst the least change occurred at cage stations with the lowest initial impact (Fig. 4.6). There were insufficient data to establish such a relationship at Creeses Mistake.

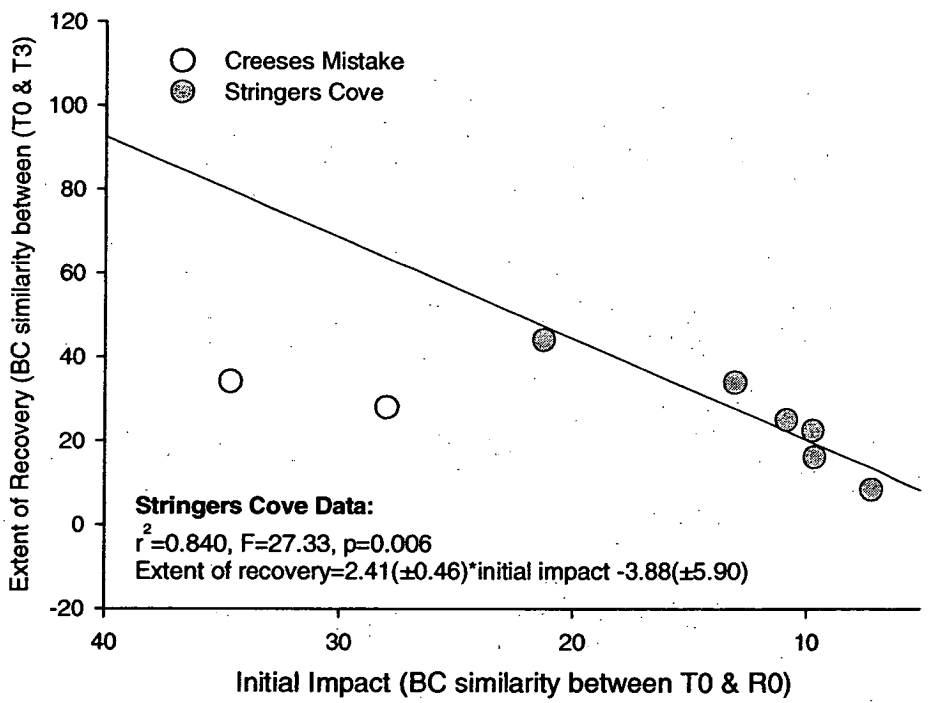

Fig. 4.6 Initial impact versus magnitude/extent of recovery over fallow period (BC similarity between T0 and T3). Regression line and equations shown are for Stringers Cove data only. Note that low initial impact levels correspond to high similarity levels (T0-R0) whilst high initial impacts are consistent with low similarity levels (T0-R0). Values in brackets are SE's of regression parameters.

The final recovered condition at the end of the three month fallow period (T3-R3) was also strongly related to the initial impact level ($r^2 = 0.70$, F = 13.72, df 1,2, p = 0.01). As the initial impact level increased the similarity between the final community structure and the reference community decreased (Fig. 4.7). An increased regression coefficient and examination of residuals indicated that this relationship was exponential. Therefore when the initial impact levels were high (i.e. < 20% similarity) there was a much greater difference between the final community structure and the reference community than at low initial impact levels (Fig. 4.7). Overall sediments at Stringers Cove had higher impact levels than at Creeses Mistake.

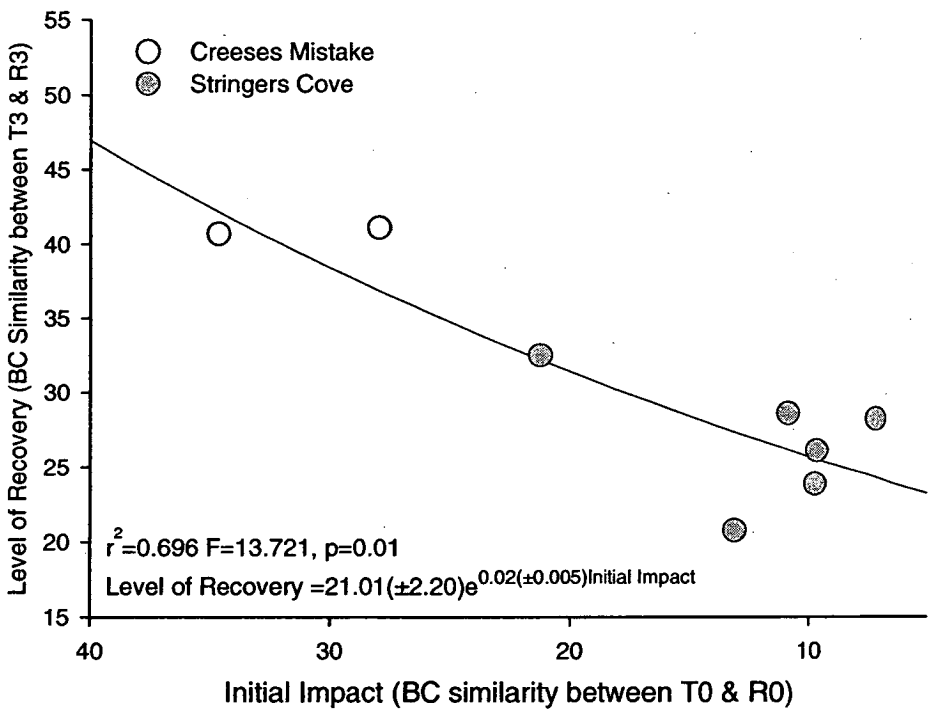

Fig. 4.7 Initial impact versus recovered status at end of fallow period as defined by the similarity of final community to reference conditions. Note that low initial impact levels correspond to high similarity levels (T0-R0) whilst high initial impacts are consistent with low similarity levels (T0-R0). Values in brackets are SE's of regression parameters.

Farming practices can have a major influence on impact level and recovery

response. Feed input was relatively consistent among cage stations farmed in the same production cycle, but differed between cycles (Table 4.1). In the second production cycle there was a marked reduction in the feed input at both farms. However, there was no evidence of any relationship between feed input and rate of recovery ($r^2 = 0.33$, F = 3.00, df = 1,6, p = 0.134), extent of recovery ($r^2 = 0.40$, F = 4.02, df = 1,6, p = 0.092) or initial impact ($r^2 = 0.37$, F = 3.556, df = 1,6, p = 0.108). All of the cage stations were subject to the additional impact of at least one adjacent cage throughout the stocked phase with several cages having 2 adjacent cages operational during the stocked phase (Table 4.1). However, there was also no evidence to suggest that adjacent cages affected the initial impact levels ($r^2 = 0.02$, F = 0.124, df = 1,6, p = 0.736).

Interestingly, there was a marked difference in the level of recovery between the different production cycles, with the greatest change occurring after the second production cycle (Fig. 4.8). On the whole, the community structure after the second production cycle returned to a state more closely resembling the pre-stocking conditions (TX-T3) (t = –3.39, df 8, p = 0.009). Although there was no significant difference in the extent of the recovery (T0-T3) between production cycles at either farm (Creeses – t = 3.25, df 2, p = 0.083; Stringers - t= –1.55, df 3, p = 0.218) there was a greater difference between the final community and the equivalent reference (T3-R3) at Creeses in the first production cycle than in the second cycle (i.e. less similarity) (Creeses – t = –13.77, df 1, p = 0.046; Stringers – t = –0.29, df 3, p = 0.791).

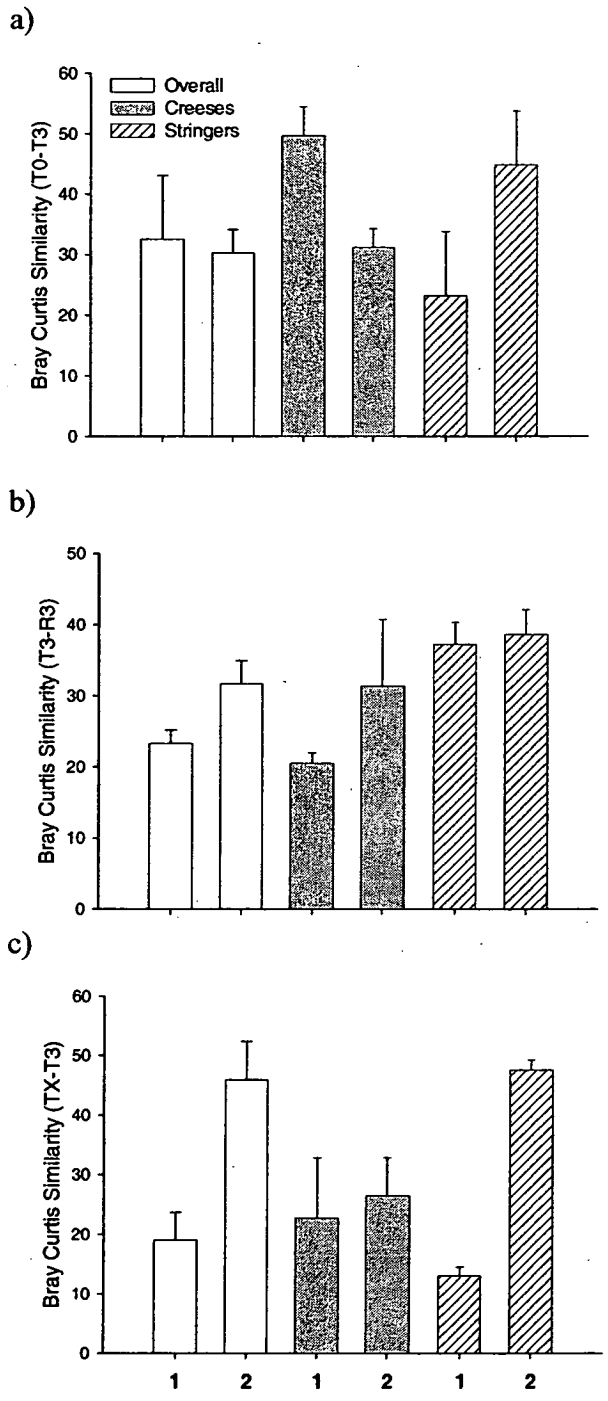

Fig. 4.8 Mean Bray-Curtis similarity level (± SE) in each production cycle (1 or 2) between a) start and end of fallow period (T0-T3), b) end of fallow period and equivalent reference (T3-R3) and c) pre-stocking and at the end of fallow period (TX-T3) for the cage communities at Creeses Mistake and at Stringers Cove.

Redox potential and sulfide concentration clearly indicated significant differences between the cage and reference sediments both immediately prior to fallowing and at the end of the three month fallow period (Redox T0 – t = –9.85, df 16, p <0.001; T3 – t = –6.09, df 16, p <0.001; Sulfide T0 – t = 2.51, df 14, p = 0.025; T3 – t = 3.24, df 14, p = 0.006) (Fig. 4.9). Redox potential was consistently lower at the cage stations than at the reference sites whilst sulfide levels were higher at the cage stations. However, over the fallow period there was no significant change in either the sulfide concentration (Creeses – t = –0.80, df 4, p = 0.470; Stringers – t = 1.302, df 7, p = 0.234) or the redox potential (Creeses – t = –1.56, df 5, p = 0.179; Stringers – t = –1.80, df 10, p = 0.101) at either farm. Physico-chemical measures of sediment condition were effective in determining impact, but were not sensitive indicators of recovery.

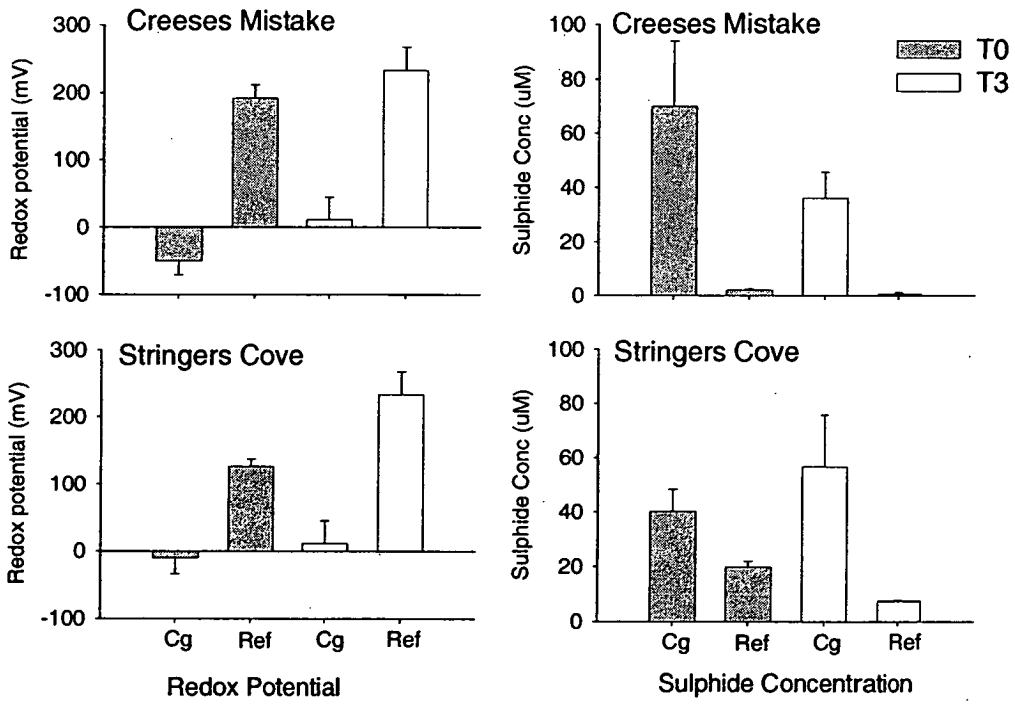

Fig. 4.9 Mean redox potential (± SE) and sulfide concentration (± SE) at cages and references at Creeses Mistake and at Stringers Cove at the start (T0) and end (T3) of the fallow period.

4.4 Discussion

In this study three levels of recovery were considered: a measurable improvement in sediment condition, a return to pre-farming condition and a return to reference condition. From the perspective of commercial aquaculture operations, recovery to pre-stocking conditions may be sufficient to sustain ongoing farming. At both farm locations the sediments at the cage stations recovered to pre-stocking levels over the three month fallow period, but they did not return to reference conditions. Despite farming practices, including the stocking levels and feed input, being similar at both farms there were marked differences in the overall recovery response and in the rate of change in the benthic community structure between the farm locations. This implies that the relationship between organic load and sediment recovery is not simple and different locations may need different fallowing strategies even when production protocols are similar. Spatial variability in recovery response has been reported in other studies. Two recent studies of salmon farms in the Broughton Archipelago, Canada showed marked differences in recovery rate. Sites in the 2004 study were estimated to have recovered in < 6 months, whereas the sites in a 2003 study were estimated to require > 6 years for recovery (Brooks *et al.* 2003, 2004).

Impact level is one of the primary factors affecting recovery response (Rosenberg *et al.*, 2002). The overall change in the community structure over the fallow period at Stringers Cove was greater than at Creeses Mistake and at the commencement of fallowing, initial impact was greater at Stringers Cove than at Creeses Mistake. These differences in impact level were not related to either feed input or the presence of adjacent cages. Local hydrographic conditions can influence impact levels and recovery rates, mitigating or exacerbating impact and recovery (Black, 2001). It has been suggested that in quiescent areas the impact may be greater and recovery may take much longer than in more hydrodynamically energetic areas (Pearson and Rosenberg, 1978, Holmer, 1991, Black, 2001). Stringers Cove was more sheltered than Creeses Mistake, suggesting that local environmental characteristics may have a major role in determining the initial impact.

At the end of the fallow period, the extent to which the farm sediments differed

from the reference conditions was broadly comparable. Consequently, even though the initial impacts differed, with equivalent time periods for recovery, the two locations returned to similar levels. This suggests that contrary to previous findings (Gowen *et al.*, 1988, Holmer, 1991 and Black, 2001), recovery was faster at the more sheltered site (Stringers Cove) than at the more exposed site (Creeses Mistake). This may be due to differences in the background ecology and natural resilience of the systems (Boesch and Rosenberg, 1981, Snelgrove and Butman, 1994, Rosenberg *et al.*, 1997). Regional differences in hydrodynamic conditions, such as wave exposure, may also affect the faunal composition (Edgar *et al.* 2005). Sheltered locations, such as Stringers Cove, tend to be naturally depositional with higher levels of organic material (Hall, 1994). Consequently, the stable state community structure of this system would reflect adaptation to a higher organic load (Boesch and Rosenberg, 1981, Llansó, 1992). This would facilitate recovery from aquaculture impacts in two ways. Firstly, there would be a natural reservoir of species able to colonise the improving sediments early in the recovery phase resulting in a more rapid return to stable state conditions and secondly because the natural conditions are already slightly organically enriched the community would attain the reference condition more quickly (Rosenberg *et al.*, 2002). Furthermore, wave or tidal disturbance may influence sediment stability at the more exposed sites impairing the ability of species to recolonise sediments and slowing recovery dynamics (Thrush *et al.*, 1992). Further investigation of the ecological relationships and function of the communities is needed to clarify the actual mechanisms involved.

At Stringers Cove the rate of change was positive signifying that similarity levels increased over time. Consequently, the greatest change in the community structure occurred in the first month, indicating that the recovery process was initially rapid but stabilized over time. In contrast, the rate of change at Creeses Mistake was negative, which indicates that each month there was a greater difference in the community structure, i.e. the initial recovery response at Creeses Mistake was slow but accelerated after the first month. Information of this kind is important for managing recovery in different systems as it indicates that changing the length of the fallow period will have a significant influence on the recovery response. Shortening the

fallow period would be likely to have a greater negative effect at Creeses Mistake where recovery was slow to start with. Consequently, it is important to understand the nature of each individual farming environment in order to manage the length of time required for recovery.

Physico-chemical parameters, such as redox and sulfide, are recommended as useful approaches for monitoring the impacts of fish farming (Hargrave *et al.*, 1997, Wildish *et al.*, 1999, Crawford *et al.*, 2002, Edgar *et al.*, 2005). Regular measurement of redox potential is currently a requirement in both the baseline and ongoing environmental monitoring programs for salmonid farms in Tasmania (Woods *et al.*, 2004). There are clearly measurable changes in sediment chemistry in the period immediately following the cessation of organic inputs (e.g. Eleftheriou *et al.* 1982, Brown *et al.*, 1987, Weston, 1990, Brooks *et al.*, 2003). In a recent study evaluating the broad-scale impacts of fish-farming in Tasmania, Edgar *et al.* (2005), specifically identified redox measured at 40 mm depth as a very sensitive indicator, able to distinguish farming effects from reference conditions. This is in agreement with the findings of chapter 2 and suggests that although redox potential and sulfide concentration appeared to be good indicators of deteriorating sediment conditions (especially those associated with major impacts) these measures returned to reference levels very quickly and as such were poor indicators of the biological condition of recovering sediments.

Although recovery rate differed between farm locations, recovery response can be predicted once the baseline environmental characteristics are understood. In order to manage sediment recovery most effectively the differences between farm locations need to be recognized and management measures tailored accordingly. Cage positions within a lease should be managed individually, initial impact levels need to be established for each cage position in order to plan for recovery and initial impact and recovery should be evaluated based on an understanding of the ecology, not just in relation to the status of the sediment chemistry or production levels. This study reinforces the findings of chapter 2 which concluded that local benchmarks are extremely important to ensure appropriate evaluation of both impact and recovery.

Chapter 5:

Biological Recovery from Organic Enrichment Associated with Finfish Cage Aquaculture: Do Some Systems Cope Better than Others?

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Macleod, C.K., Moltschaniwskyj, N.A., Crawford, C.M. and Forbes, S.E. Biological Recovery from Organic Enrichment associated with Finfish Cage Aquaculture: Do Some Systems Cope Better than Others?

Abstract

This study examined the short-term recovery response at two salmon farms in southern Tasmania where the organic loads were equivalent, but where the background environmental conditions differed markedly. Although the benthic communities at each of the farm locations showed good recovery over the three month fallow period, the community structure did not fully return to that observed under reference conditions at either location. At the Stringers Cove site the primary ecological functions of the natural community were restored, but this was not the case at Creeses Mistake. These differences in recovery response were a direct reflection of the background environmental conditions. Stringers Cove sediments had a naturally high organic carbon content and as a result there was greater similarity in the ecological function of the unimpacted and impacted conditions at this location than at Creeses Mistake where, under natural conditions, the sediments had a very low organic content. In addition, the background fauna at Stringers Cove contained several species with reproductive strategies that were suited to rapid recruitment and were well adapted for early recolonisation in organically enriched sediments. In contrast, the natural fauna at Creeses Mistake not only changed more with the impacts of organic enrichment, but was less able to re-establish their populations directly by immigration, needing to rely to a greater extent on remediation of the sediments by transitional species before being able to colonise. This has important implications for environmental management, as it suggests that the sediments in some areas have a greater natural resilience to organic inputs.

Keywords: Benthic recovery, ecological function, recolonisation, organic enrichment, resilience, macrofauna, salmon farming

5.1 Introduction

The rate and extent of recovery of soft-sediment habitats following eutrophication events is affected by many different environmental factors, both within the impacted area and on a broader scale (Thrush and Whitlach, 2001). The nature and extent of eutrophication as well as local hydrodynamic conditions will influence recovery rate and particularly affect sediment stability and faunal recruitment. Meanwhile, the natural background ecology will dictate the availability of colonists. Disturbed areas recover to contain assemblages very similar to those in the adjacent ambient community (Bonsdorff, 1989), so the composition of the background community is critical to the recovery response. A variety of physical and ecological factors influence how communities are structured: sediment grain-size, sediment stability, organic content, availability of food and trophic interactions are amongst the most important (Snelgrove and Butman, 1994), but the broader environmental factors become increasingly important as the spatial scale of comparison increases (Whitlach and Zajac, 1985). In order to fully understand the recovery response it is important to identify the key processes influencing broad-scale differences (Thrush and Whitlach, 2001).

Cage finfish aquaculture produces organic waste products (fish faeces and excess feed) that may result in localised organic enrichment of the sediments (Willoughby, 1999). Many studies have been undertaken to determine the length of time required for complete recovery after fish farming, and there is considerable variability in the estimates, ranging from months (Gowen *et al.*, 1988, Lumb, 1989, Ritz *et al.*, 1989, Johannessen *et al.*, 1994, Pohle *et al.*, 2001, Pereira *et al.*, 2004) to years (Karakassis *et al.*, 1999, Brooks *et al.*, 2004). It is generally agreed that finfish cage aquaculture affects the infaunal community in the same way as other sources of organic enrichment. A confounding factor is that most ongoing cage fish farming operations involve repeated organic enrichment events, based around the individual management requirements of the farming operations which often involves an annual or seasonal rotation of cages within an area. In terms of biogeochemical and infaunal response this may have more similarity to natural organic enrichment events (e.g.

fluctuations in river discharges or oceanic nutrient loads) than to other anthropogenic impacts which tend to be either relatively constant (e.g. sewage outfalls or wood pulp mill effluent) or episodic (e.g. oil spills). There is very little information on the effects of such short-term fallowing or cage rotation on sediment recovery.

The amount of time required for sustainable sediment remediation is at present poorly understood. For ongoing farming operations to be sustainable it may not be necessary for the sediments to fully recover after each production cycle. In determining a suitable recovery level for sediments subject to recurrent impact it may be more appropriate to determine whether sediments have recovered sufficiently to be able to withstand further inputs without suffering any cumulative deterioration. If fallowing protocols fail to return sediments to such a condition, then there is a danger that sediment condition may progressively deteriorate to such an extent that ecological function is significantly impaired and farming operations may become unviable. Consequently, the sustainability of ongoing and repetitive impacts, such as fish farming may be better assessed by establishing whether the ecological function of the system has been restored.

Most studies on the recovery of cage aquaculture operations have focussed on recovery at single sites/leases within similar geographic areas (Ritz *et al.*, 1989, Karakassis *et al.*, 1999, Pereira *et al.*, 2004) and there are very few aquaculture-based studies that specifically compare the recovery responses of geographically distinct locations. Ecological conditions can have significant affects on the environmental sustainability of farming operations. Consequently, it might be expected that the sediments at different farm locations, with differing environmental conditions, would respond quite differently to similar fallowing protocols. Brooks *et al.* (2004) found significant differences in recovery rates between 2 farm leases within the same region in Canada and suggested that this may be linked to changes in depositional nature of the environment. Similarly in reviewing the findings of several aquaculture recovery studies Black (2001) attributed differences in the overall estimates of recovery time to broad scale variability in the underlying environmental conditions. Finfish farming operations in the temperate waters of Tasmania are located in a range of environments. In a recent study examining indicators of farm effects at 20 separate farm lease

locations in Tasmania, Edgar *et al.* (2005) suggested that there were marked differences in environmental conditions and that exposure level may be a major determinant of regional variability in background ecology. A comparison of the differences in short-term recovery response, particularly changes in functional ecology, at farms with differing levels of environmental exposure would enable aquaculture and environmental managers to more accurately assess the sustainability of cage farming.

In the previous chapter it was established that impact level was a significant factor in determining recovery response and that local environmental conditions may have a major role in determining both impact and rate of recovery. This study expands on the findings in Chapter 4 to examine broad scale differences in the ecological responses of infaunal communities to short-term periods of recovery from organic enrichment. In addition, the management implications of regional variability in ecological function of the infaunal community in relation to sediment recovery were assessed.

5.2 Methods

5.2.1 Study sites

Two farm locations in south eastern Tasmania, Creeses Mistake (Tasman Peninsula) and Stringers Cove (Port Esperance) were sampled in this study (Fig. 5.1). These farms are broadly representative of the differing environments in which Atlantic salmon culture is undertaken in Tasmania. Creeses Mistake is a relatively exposed, shallow (20 m water depth) and fully marine site with predominantly fine sand sediments, whereas Stringers Cove is in deeper (40 m) more sheltered waters that are occasionally subject to the freshwater influence of the nearby Esperance River with finer silt/clay sediments (Macleod *et al.* 2002).

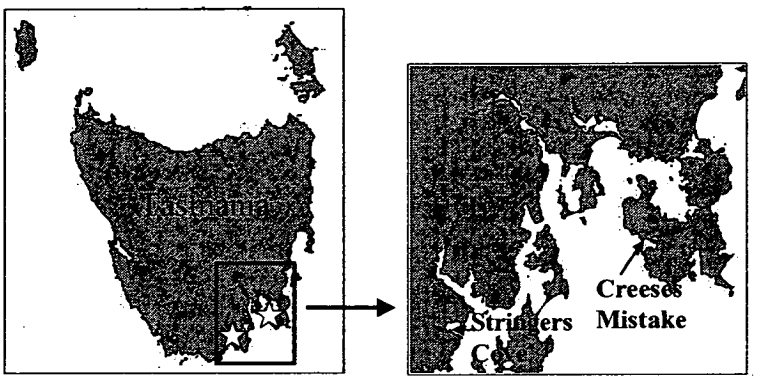


Fig. 5.1 Location of study farms in south eastern Tasmania. Creeses Mistake is located in Wedge Bay on the Tasman Peninsula and Stringers Cove is within Port Esperance.

Changes in the community structure associated with standard farm fallowing protocols were studied over two annual production cycles. Both farms employed an annual stocking regime where cages were stocked for nine months and then fallowed for three months. Sediment samples were collected from cage stations and references prior to the cages being stocked (TX), at the end of nine months of stocking (i.e. at the start of the fallow period - T0), and at the end of a three month fallow period (T3). During the second year additional samples were collected from cage and reference stations at monthly intervals during the fallow period. The data analysed in this chapter represent a subset of the data used in Chapter 4.

Prior to sampling, each farm area was mapped using a Garmin 135 GPS Map unit coupled with a Racal differential unit. Depth and positional information were collected for all cages present on the lease at the time. In addition, reference locations, within the same depth range, but 150 m distant from the edge of selected study cages, were located using the depth contours and GPS.

5.2.2 Faunal sampling

Five replicate sediment samples were collected from each cage station and reference using a Van Veen Grab (surface area – 0.0675 m^2) (Fig. 5.2). Grab contents were transferred to mesh bags (mesh size 0.875 mm) and rinsed. Samples were wet seived to 1 mm and the retained material preserved in a solution of 10%

formalin:seawater (4% formaldehyde). Samples were transferred to the laboratory for sorting and the infauna identified to the lowest possible taxonomic level and enumerated.

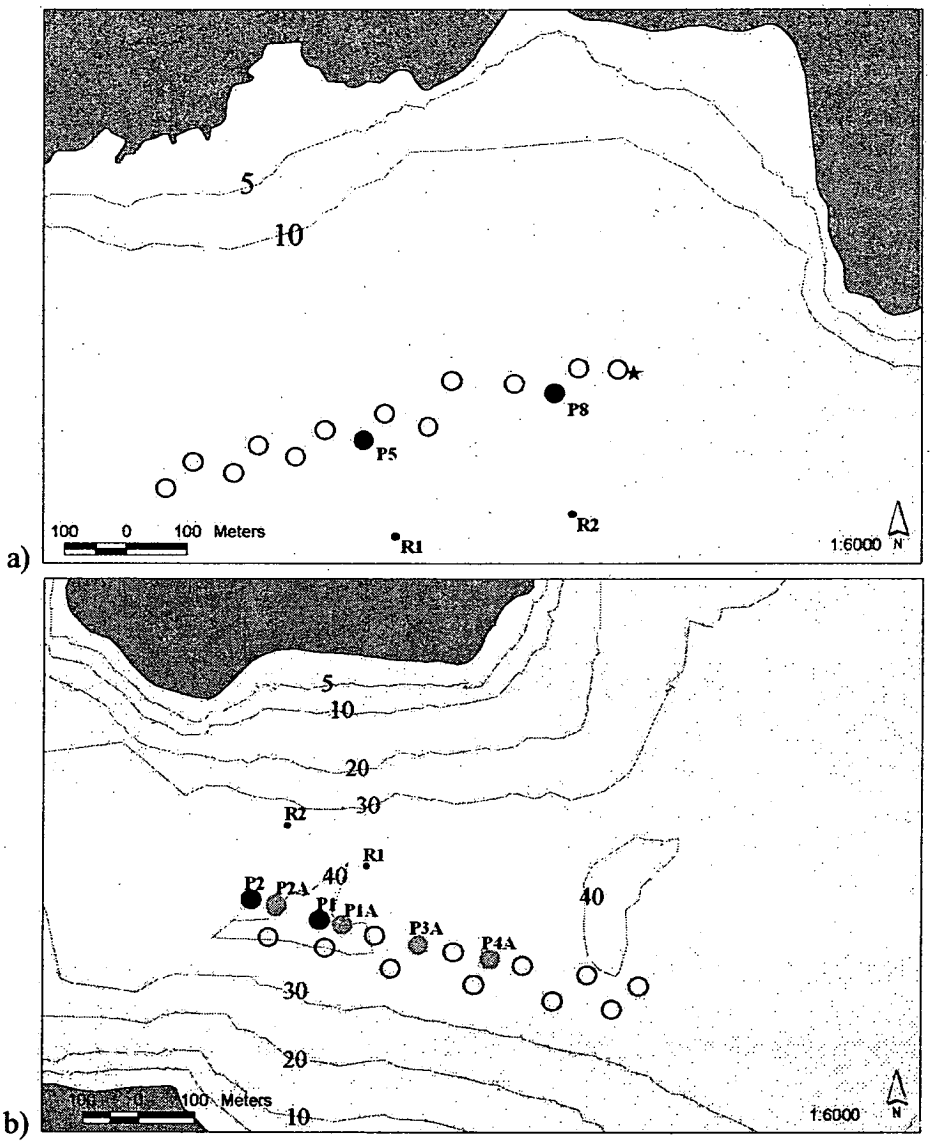


Fig. 5.2 Cage stations and reference sites for sediment samples for a) Creeses Mistake and b) Stringers Cove. Stringers Cove sample sites shown as ● were stocked in the 1st production cycle whilst those shown as ◍ were stocked in the 2nd cycle.

5.2.3 Statistical analysis

Univariate statistical analyses were used to examine spatial and temporal differences in the community structure. Analysis of variance (ANOVA) was used to compare abundance, number of species, and Shannon diversity both between farm locations, and within farms over the fallow period.

A similarity matrix was derived from abundance data using Bray-Curtis similarity indices and this was then used to determine the relationships between the community structures at the replicate cages and references. Relationships between references and cages at each of farms over the fallow period were displayed as an ordination plot using non-metric multidimensional scaling (nMDS) to identify faunal relationships. Abundance data were square root transformed to reduce the influence of abundant taxa and the Bray-Curtis similarity index was used because of the robustness of this statistic to zero-inflated data sets (Clarke, 1993). The significance of differences between conditions prior to stocking, at the reference stations and at the end of the fallow period was assessed using planned contrasts and the ANOSIM randomisation test. SIMPER analysis was used to determine the relative contribution of each taxa to the average similarity within groups and dissimilarity between groups.

Key ecological and reproductive strategies were defined for the ten dominant species at the references and at the cage stations prior to (T0) and post- (T3) fallowing. The main ecological and life history classifications were feeding strategy, role in sediment, and reproductive strategy. Three main feeding strategies were identified: predatory carnivores (C), suspension feeders (SF), or deposit feeders (DF). However, many animals change their feeding strategies in response to environmental conditions and where this was deemed to be the case they were classified as having a variable feeding strategy (VF). The primary role that the animals play with respect to the sediments was defined as either sediment bioturbators/ destabilisers (SD) or consolidators (SC). The main reproductive strategy of each species was characterised by whether adults had staged/seasonal (SR) or opportunistic (OR) reproductive cycles and whether larvae were benthic (B), pelagic (P), a mix of both (B/P), or the reproductive strategy was unknown (UR). Australian marine and estuarine infauna is on the whole relatively poorly described, with very little species-specific biological and ecological information. Where the ecology/biology was not specifically known it has

been inferred using information from the next closest species or taxonomic group. Principal components analysis (PCA) of the ecological data was used to reveal differences between cage and reference stations, with the associated biplots showing the major faunal groups, sediment and feeding role and reproductive strategies most responsible for the community changes.

All multivariate analyses were conducted using the Plymouth Routines in Multivariate Ecological Research (PRIMER) software package (PRIMER 2006).

5.3 Results

There were significant differences between the community structures of the reference and cage stations at each of the two farm locations (ANOSIM Rho = 0.577, $p < 0.001$) (Fig. 5.3). The reference communities in particular showed major differences between the farms (85% Bray-Curtis dissimilarity) (ANOSIM Rho = 1.000, $p < 0.001$). However, there was still considerable overlap in species composition: 27% of species were common to both farm reference sites, with a relatively even proportion of these between annelids (33%), crustaceans (22%), molluscs (29%) and echinoderms (12%). There were no numerically important species that were unique to the community at either farm location. so that differentiation of the populations at the 2 farms was a result of changes in relative abundance within a suite of species. However, although the overall combination of species was similar at both locations, the dominant taxa were markedly different (Fig. 5.4 and 5.5).

The reference communities at Creeses Mistake had significantly more individuals and species than those at Stringers Cove (ANOVA abundance $F = 17.36$ df 1, 29, $p < 0.001$, diversity $F = 38.83$, df 1, 29, $p < 0.001$) (Fig. 5.4). There was an average of 109 (± 12) species at Creeses Mistake compared with only 70 (± 6) at Stringers Cove and the abundance was three times greater at Creeses Mistake (4,085 (± 731)) than at Stringers Cove (1,370 (± 98)). At Creeses Mistake the communities at the reference sites were characterised by a range of species from several different taxa (Table 5.1), but crustaceans accounted for the largest component of the fauna (82% of the abundance and 52% of species) (Fig. 5.4). Eight of the top 10 dominant species

were crustaceans (Table 5.1). The crustaceans *Apseudes* sp.2, *Ampelisca* sp., *Euphilomedes* sp. and *Birubius* sp. were all abundant and important species characterizing the Creeses Mistake reference sites (Table 5.1). In contrast, at Stringers Cove annelids were the dominant taxon at the references comprising 46% of the faunal abundance and 41% of the species (Fig. 5.4). At Stringers Cove the polychaete *Mediomastus australiensis* and the brittlestar *Amphiura elandiformis* were the dominant species in the reference fauna (Table 5.2).

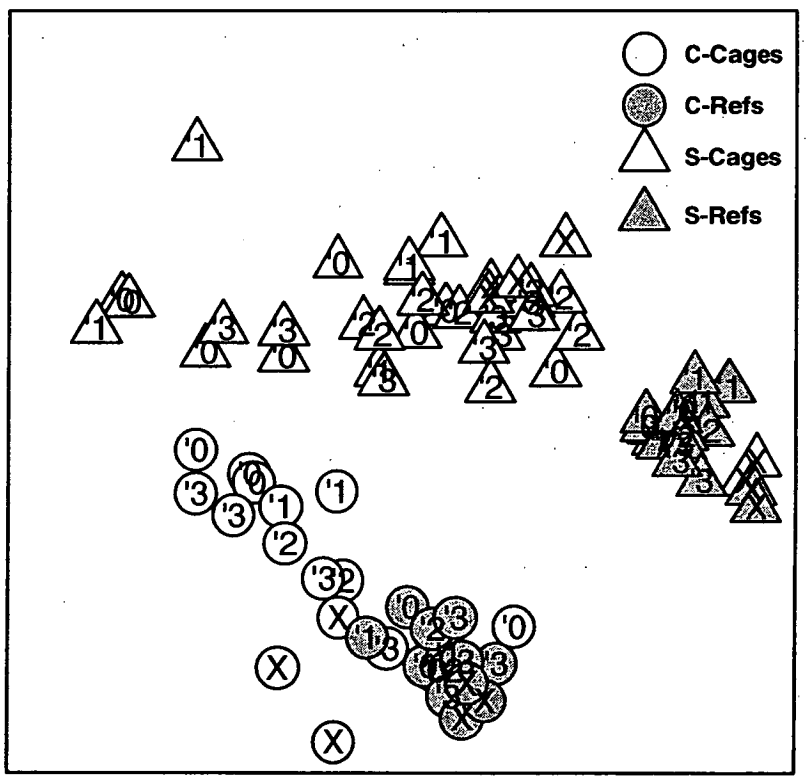


Fig. 5.3 Ordination of Bray-Curtis similarity matrix for reference and cage stations pre-stocking (X), at the start of the fallow period (0) and at 1, 2 and 3 monthly intervals over the fallow period. Data were square root transformed. Stress=0.16.

Table 5.1 The relative contribution to group similarity, abundance, proportional representation within community group, ecological function and reproductive strategies of the ten species which contributed most to the overall group similarity of the reference, pre-fallowing (T0) and post-fallowing (T3) communities at Creeses Mistake. Ecological function has been derived from literature sources. Where species specific information was unavailable the nearest taxonomically similar species/group has been used, these results are indicated in italics. Key to Ecological Function: LG-Large bodied animal (likely to be retained on a 10 mm sieve), MD-Medium sized animal (retained on a 4mm sieve), SM- small animal (retained on a 1mm sieve), SF- Suspension feeder, DF-Deposit Feeder, S-Sediment stabiliser, D-Sediment Destabiliser. Key to Reproductive Strategy: MG-Adults produce multiple generations, SG- adults spawn only once, P-Planktonic juveniles, B-Benthic juvs, SR-Staged reproductive timing (episodic/seasonal), OR- Opportunistic reproductive cycle.

Reference (Average similarity 48.85)

Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Apseudes sp.2	Crustacea	11.48	662 (250.1)	*20.5*	*MD: DF; D*	*MG:B:SR*
Ampelisca sp.	Crustacea	9.79	455 (148.4)	*14.0*	*SM: SF; S*	*SG:B:SR(2)*
Euphilomedes sp.(MoV18)	Crustacea	7.29	224 (45.7)	*6.9*	*SM: DF; D*	*Unknown*
Birubius sp.	Crustacea	6.81	161 (29.6)	*5.0*	*SM: DF; D*	*MG:B:SR*
Corophium ascherusicum	Crustacea	4.07	145 (37.4)	*4.5*	*SM: DF/SF: S*	*MG:B:SR/OR*
Lysianassidae sp.4	Crustacea	3.03	98 (23.4)	*3.0*	*MD: C: S*	*MG:P:SR*
Echinocardium cordatum	Echinodermata	2.92	73 (19.88)	2.3	LG: DF; D	MG:P:SR
Phyllamphicteis sp.(MoV3094)	Annelida	2.76	75 (22.8)	*2.3*	*LG: SF/DF:S*	*Unknown*
Protolembos sp.	Crustacea	2.73	52 (16.2)	*1.6*	*SM: SF/DF:S*	*MG:B:SR*
Nebalia longicornis	Crustacea	2.71	36 (8.5)	*1.1*	*MD: DF: D*	*MG:P:OR*

Pre-fallowing (T0) (Average similarity 62.84)

Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Capitella capitata (complex)	Annelida	73.12	17,248 (667.2)	93.1	SM: DF: D	MG,B/P,OR
Neanthes cricognatha	Annelida	4.24	199 (113.9)	*1.1*	*MD: DF/C: D*	*MG:B:SR*
Maladnidae sp.	Annelida	2.83	54 (24.4)	*0.3*	*MD: DF: S*	*SG;B;SR*
Mysella donaciformis	Mollusca	2.06	14 (6.2)	*0.1*	*MD-LG: DF/SF: D*	*MG;B:SR*
Euphilomedes sp.(MoV18)	Crustacea	1.70	52 (32.3)	*0.3*	*SM: DF: D*	*Unknown*
Corbula gibba	Mollusca	1.53	11 (6.5)	0.1	LG: DF: D	MG:P:SR
Ostracoda sp.	Crustacea	1.33	10 (4.5)	*0.1*	*SM: DF: D*	*Unknown*
Nebalia longicornis	Crustacea	1.27	57 (31.1)	*0.3*	*MD: DF: D*	*MG:P:OR*
Apseudes sp.2	Crustacea	1.05	21 (9.5)	*0.1*	*MD: DF: D*	*MG:B:SR*
Birubius sp.	Crustacea	1.03	65 (35.6)	*0.4*	*SM: DF: D*	*MG:B:SR*

Post-fallowing (T3) (Average similarity 35.94)

Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Capitella capitata (complex)	Annelida	26.38	2621 (1343.8)	74.8	SM: DF: D	MG,B/P,OR
Amphithoidae sp.	Crustacea	6.78	37 (14.6)	*1.1*	*SM: SF/DF:S*	*MG:B:SR*
Euphilomedes sp.(MoV18)	Crustacea	5.47	73 (38.0)	*2.1*	*SM: DF: D*	*Unknown*
Birubius sp.	Crustacea	4.82	36 (23.0)	*1.0*	*SM: DF: D*	*MG:B:SR*
Solemya australis	Mollusca	4.72	36 (19.4)	*1.0*	*LG: DF/SF: S*	*MG:P:Unknown*
Oedicerotidae sp.	Crustacea	4.26	25 (9.7)	*0.7*	*SM: DF:D*	*MG:B:SR*
Neanthes cricognatha	Annelida	4.00	94 (55.5)	*2.7*	*MD : DF/C: D*	*MG:B:SR*
Apseudes sp.2	Crustacea	3.66	21 (9.9)	*0.6*	*MD: DF: D*	*MG:B:SR*
Ostracoda sp.	Crustacea	3.45	67 (50.8)	*1.9*	*SM: DF: D*	*Unknown*
Nassarius nigellus	Mollusca	2.57	22 (10.0)	*0.6*	*MD: DF: E*	*MG:B:SR*

Table 5.2 The relative contribution to group similarity, abundance, proportional representation within community group, ecological function and reproductive strategies of the ten species which contributed most to the overall group similarity at the reference, pre-fallowing (T0) and post-fallowing (T3) communities at Stringers Cove. Ecological function has been derived from a variety of literature sources; where no species specific information available the nearest taxonomically similar species/group was used, these results are indicated in italics. Key to Ecological Function: LG-Large bodied animal (likely to be retained on a 10mm sieve), MD-Medium sized animal (retained on a 4mm sieve), SM- small animal (retained on a 1mm sieve), SF- Suspension feeder, DF-Deposit Feeder, S-Sediment stabiliser, D-Sediment Destabiliser. Key to Reproductive Strategy: MG-Adults produce multiple generations, SG- adults spawn only once, P-Planktonic juveniles, B-Benthic juvs, SR-Staged reproductive timing ((1)-once a year, (2)- episodic/seasonal/twice a year), OR- Opportunistic reproductive cycle.

References (Average similarity 63.24)

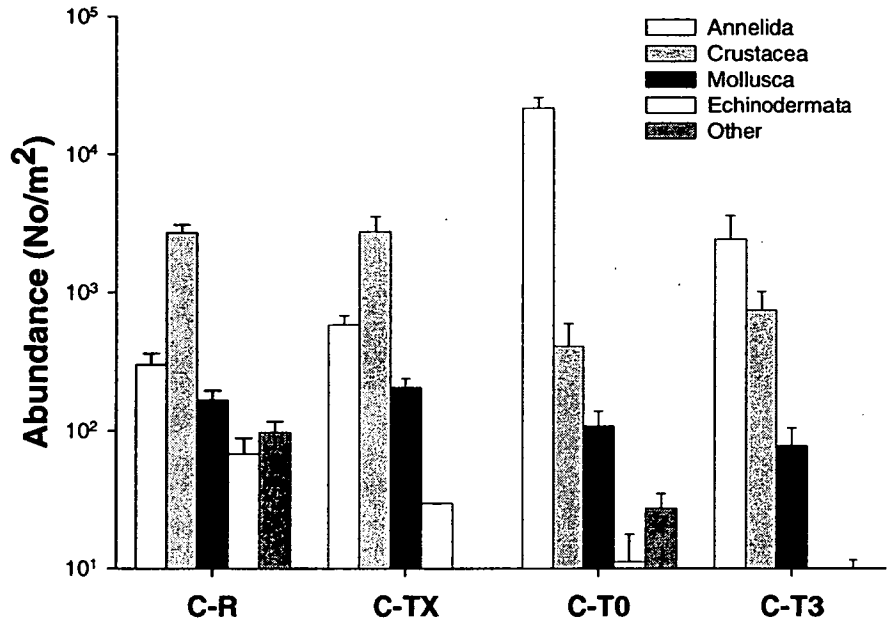
Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Mediomastus australiensis	Annelida	10.17	213 (32.3)	*14.8*	*MD: DF: D*	*MG,B/P,SR*
Amphiura elandiformis	Echinodermata	8.19	128 (9.7)	*8.9*	*LG: DF/SF: D*	*MG:P:SR*
Lysilla jennacubinae	Annelida	5.98	75 (11.0)	*5.2*	*LG: DF: S*	*Unknown*
Nucula pusilla	Mollusca	5.97	121 (25.3)	*8.4*	*LG: DF: D*	*MG:P:SR*
Thyasira adelaideana	Mollusca	5.01	58 (4.5)	*4.0*	*LG: SF: D*	*MG:B:SR*
Nassarius nigellus	Mollusca	4.48	94 (22.9)	*6.5*	*MD: DF: E*	*MG:B:SR*
Aricidea sp.	Annelida	4.37	75 (16.3)	*5.2*	*MD: DF: D*	*SG;P;SR*
Nemertea sp.1	Nemertea	4.25	64 (15.9)	*4.4*	*MD: C/DF: D*	*MG:B:SR*
Lumbrinereis sp.1	Annelida	4.07	38 (5.7)	*2.7*	*LG: DF:D*	*SG;B;SR*
Aschyis sp.2	Annelida	3.91	78 (25.1)	*5.4*	*LG: DF: S*	*SG;B;SR*

Pre-fallowing (T0) (Average similarity 39.37)

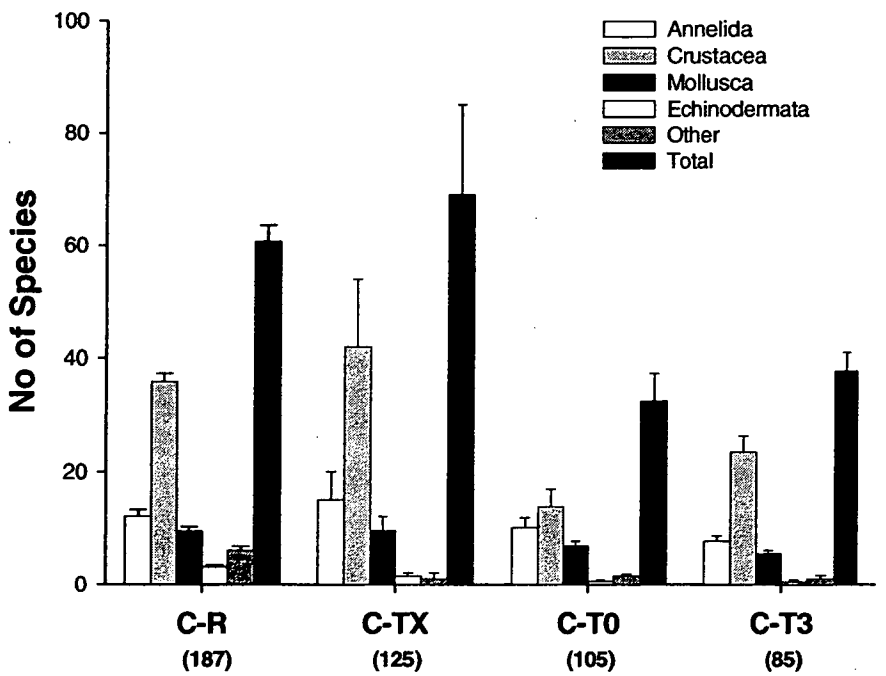
Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Nebalia longicornis	Crustacea	35.85	14,902 (7,051)	*34.2*	*MD: DF: D*	*MG:P:OR*
Capitella capitata (complex)	Annelida	22.71	27,470 (12,806)	63.1	SM: DF: D	MG,B/P,OR
Corbula gibba	Mollusca	7.17	353 (149.2)	0.8	MD-LG: DF: D	MG:P:SR
Birubius spp.	Crustacea	6.41	201 (79.5)	*0.5*	*SM: DF: D*	*MG:B:SR*
Nassarius nigellus	Mollusca	5.53	155 (87.3)	*0.4*	*MD: DF: E*	*MG:B:SR*
Neanthes cricognatha	Annelida	4.29	118 (53.4)	*0.3*	*SM-MD: DF/C: D*	*MG:B:SR*
Echinocardium cordatum	Echinodermata	3.35	43 (15.4)	0.1	LG: DF: D	MG:P:SR
Simplisetia amphidonta	Annelida	2.13	8 (1.3)	*<0.1*	*MD: DF/C: D*	*SG:P:SR*
Nemertea sp.1	Nemertea	1.54	4 (4.8)	*<0.1*	*MD: C/DF: D*	*MG:B:SR*
Mediomastus australiensis	Annelida	1.30	9 (3.5)	*<0.1*	*MD: DF: D*	*MG,B/P,SR*

Post-fallowing (T3) (Average similarity 42.21)

Species ID	*Phylum*	*Simil Contrib%*	*No/m²*	*Proportion of Total*	*Ecological Function*	*Reproductive Strategy*
Nassarius nigellus	Mollusca	19.07	201 (42.0)	*2.5*	*MD: DF: E*	*MG:B:SR*
Birubius spp.	Crustacea	13.09	101 (23.1)	*1.2*	*SM: DF: D*	*MG:B:SR*
Corbula gibba	Mollusca	11.66	222 (77.5)	2.7	MD-LG: DF: D	MG:P:SR
Echinocardium cordatum	Echinodermata	9.61	71 (21.6)	0.9	LG: DF: D	MG:P:SR
Capitella capitata (complex)	Annelida	8.79	5,525 (3,685)	67.7	SM: DF: D	MG,B/P,OR
Theora fragilis	Mollusca	6.09	23 (6.2)	*0.3*	*MD: DF: D*	*MG:P:SR*
Nebalia longicornis	Crustacea	3.76	1,791 (1,319)	*22.0*	*MD: DF: D*	*MG:P:OR*
Paraprionospio coora	Annelida	3.05	11 (3.7)	*0.1*	*MD: DF: D*	*SG:P:Unknown*
Mysella donaciformis	Mollusca	3.01	11 (3.7)	*0.1*	*MD-LG: DF/SF: D*	*MG;B:SR*
Simplisetia amphidonta	Annelida	2.93	10 (2.6)	*0.1*	*MD: DF/C: D*	*SG:P:SR*



a)



b)

Fig. 5.4 a) Average abundance (s.e.) and b) number of species (s.e.) in each of the main faunal groups at Creeses Mistake at the references at the start of fallowing and at the cage stations prior to stocking (TX), and at the start (T0) and end (T3) of the fallow period. The total number of species recovered is indicated in parenthesis beneath the site labels (X-axis) on Fig. b.

At Creeses Mistake the pre-stocking community was not significantly different from the reference community (ANOSIM Rho = 0.500, p = 0.33) (Fig. 5.3). However, the fauna changed markedly with farming such that at the start of the fallow period (T0) the community structure was very different to both the references (ANOSIM Rho = 1.000, p = 0.03) and the pre-stocking community (ANOSIM Rho = 1.000, p = 0.07) (Fig. 5.3). After farming, the cage stations (T0) were dominated by annelids (97% of fauna) and remained this way until the end of the fallow period (T3) (Fig. 5.4). The change to annelid dominance was also evident in the number of species, although this was not quite so marked (Fig. 5.4). The overall number of species at the cage stations declined from 71 to just 33, with a much greater proportional decline in the crustacean (42 to 14 species) compared to the annelid (15 to 10 species) fauna (Table 5.1). At the start of fallowing the polychaete *Capitella capitata* complex was present in excess of 17,000 m^{-2} and alone accounted for 73% of the overall similarity within the community (Table 5.1). Although this had declined to 2,620 m^{-2} and 26% at the end of the three month fallow period, it was still markedly more abundant in the post-stocking community than it had been pre-stocking (232 m^{-2}) (Table 5.1).

Conditions appeared to improve over the fallow period, with the final community closer to that found prior to stocking (ANOSIM Rho = 0.545, p = 0.05), however, the community structure was still significantly different to the references at the end of the fallow period (ANOSIM Rho = 0.728, p = 0.01) (Fig. 5.3). The main difference between the pre- and post-fallowing communities was as a result of changes in abundance of three key polychaete species. In the post-fallowing communities *Capitella capitata* complex declined tenfold from 21,344 m^{-2} to 2,621 m^{-2}, *Neanthes cricognatha* declined from 199 m^{-2} to 94 m^{-2} and Maldanidae sp. was absent altogether (Table 5.3). The proportional importance of crustaceans increased over the fallow period, and although crustaceans were still markedly less abundant after fallowing than in either the pre-stocking or reference communities (Fig. 5.4), there was clearly some recovery in this group.

Table 5.3 Differences in abundance and Bray Curtis dissimilarity level of the ten species which most clearly distinguish between pre- (T0) and post-fallowing (T3) communities and between the pre-stocking (TX) and post-fallowing (T3) communities at Creeses Mistake.

Species ID	*T0 (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissimil. %*
Capitella capitata complex	17,248 (6,667)	2,621 (1,344)	3.48	33.18
Neanthes cricognatha	199 (133.9)	94 (55.5)	1.71	2.92
Maldanidae sp.	54 (24.4)	0	2.82	2.19
Spionidae sp	6 (6.2)	117 (71.6)	0.86	2.03
Tethygeneia sp.(MoV 1304)	0	14 (13.6)	0.66	1.89
Corophium ascherusicum	115 (53.2)	6 (4.7)	1.19	1.88
Amphithioidae sp.	10 (7.0)	37 (14.6)	1.74	1.66
Nebalia longicornis	57 (31.1)	23 (20.3)	1.28	1.62
Cumacea sp.	1 (1.2)	2 (2.5)	0.60	1.52
Oedicerotidae sp.	1 (1.2)	25 (9.7)	1.77	1.46

Species ID	*TX (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissimil. %*
Capitella capitata complex	232 (74.8)	2,621 (1,343.8)	1.18	7.63
Apseudes sp.2	530 (507.9)	21 (9.9)	1.08	5.39
Birubius spp.	474 (456.9)	36 (23.0)	1.22	5.22
Spionidae sp.	122 (69.9)	0	3.72	3.57
Lyssianassidae sp.1	176 (126.6)	6 (3.1)	2.54	3.27
Tethygeneia sp.(MoV 1304)	232 (156.7)	14 (13.6)	1.85	2.23
Halicarcinus ovatus	102 (92.4)	2 (1.4)	1.2	2.07
Spionidae sp	0	117 (71.6)	0.76	1.60
Solemya australis	77 (44.4)	36 (19.4)	1.8	1.59
Theora fragilis	26 (17.4)	0	5.07	1.56

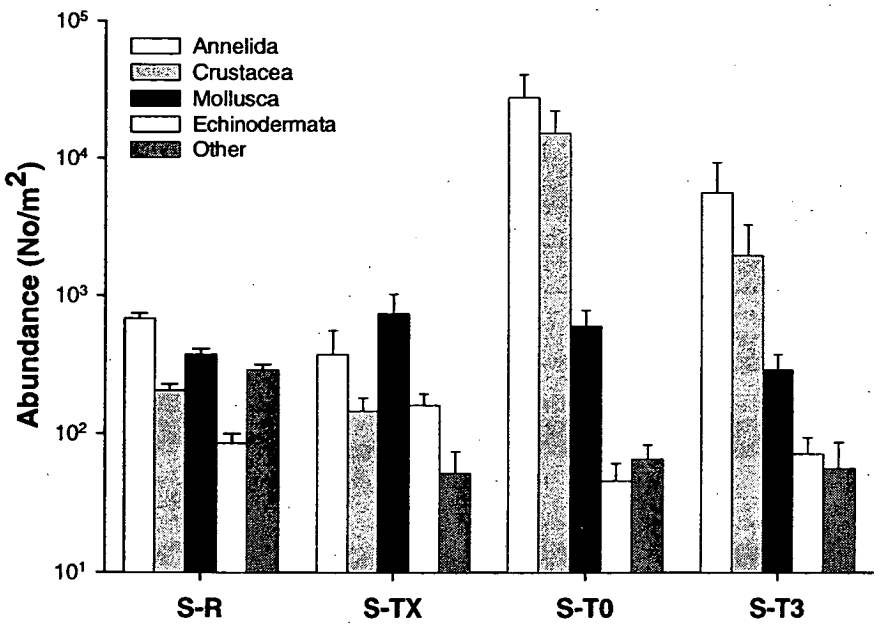
Species ID	*R3 (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissimil. %*
Capitella capitata complex	5 (3.5)	2,621 (1,344)	1.33	12.39
Ampelisca sp.	414 (255.7)	5 (3.5)	2.17	4.43
Apseudes sp.2	607 (399.9)	21 (9.9)	1.38	4.36
Spionidae sp	1 (1.23)	117 (71.6)	1.01	2.21
Euphilomedes sp (MoV18)	215 (43.0)	73 (38.0)	1.33	2.17
Lyssianassidae sp.4	78 (20.8)	23 (23.5)	1.73	2.12
Photis sp.	100 (59.6)	4 (3.7)	1.81	2.06
Neanthes cricognatha	4 (2.4)	94 (55.5)	1.05	2.05
Echinocardium cordataum	65 (32.8)	0	2.10	2.02
Corophium ascherusicum	102 (48.9)	6 (4.7)	1.71	2.02

Although the total abundances at Stringers Cove were similar in the pre-stocking and reference communities (ANOVA $F = 0.002$ df 1, 20 $p = 0.969$) the number of species in each of the major faunal groups was consistently and significantly lower in the pre-stocking communities (ANOVA $F = 9.917$ df 1, 20 $p = 0.005$) (Fig. 5.5). There were significant differences in the species composition between the reference, pre-stocking, pre-fallowing and post-fallowing communities (ANOSIM Rho

= 0.437, p = 0.001). The Rho values from pairwise comparison of the data were lower for the pre-stocking comparison indicating that the pre-stocking community (TX) was closer to the post-fallow community (T3) (ANOSIM Rho = 0.183, p = 0.04) than to either the reference (ANOSIM Rho = 0.542, p = 0.04) or pre-fallow communities (T0) (ANOSIM Rho = 0.467, p = 0.01). Annelids comprised the greatest proportion of the reference community (37% of total abundance, 53% of species; Fig. 5.5). The pre-stocking community still had proportionally many more species of annelid than any other taxonomic group (44%) but the abundance of molluscs had increased markedly (Fig. 5.5), mostly due to increases in the abundance of *Corbula gibba* (from 37 m^{-2} to 515 m^{-2}) and *Nassarius nigellus* (89 m^{-2} to 151 m^{-2}) at the pre-stocking stations.

At the start of fallowing (T0) the abundance of the two species *Capitella capitata* complex, a polychaete, and *Nebalia longicornis*, a crustacean, had increased markedly (Table 5.4). The large increase in abundance of these two species was the primary cause of the significant increase in annelids (ANOVA F = 4.301 df 1, 22 p = 0.01) and crustaceans (ANOVA F = 4.301 df 1, 22 p = 0.005) relative to the references at this time (Fig. 5.5). Over the fallow period the abundance of these species declined markedly (Table 5.4), but at the end of three months fallowing (T3) the community structure remained significantly different from the reference (ANOSIM Rho = 0.759, p = 0.01). Several molluscs, particularly the scavenging gastropod *Nassarius nigellus* and the introduced bivalves *Corbula gibba* and *Theora fragilis*, were important features of the fallowed communities (Table 5.2). However, the most important species change over the fallow period was the decline in abundance of the opportunistic species *Capitella capitata* complex and *Nebalia longicornis* (Table 5.4).

a)



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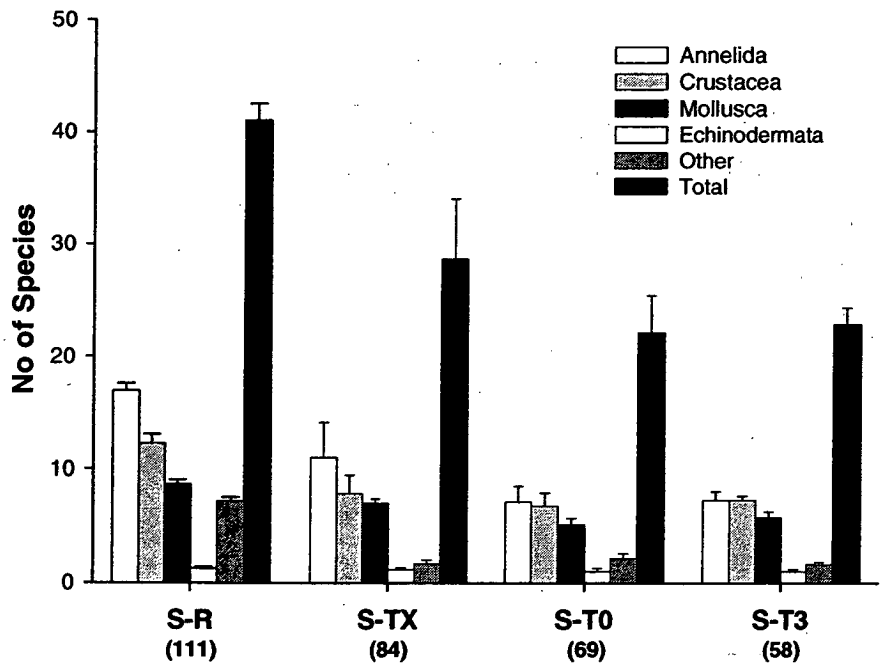


Fig. 5.5 a) Average abundance (s.e.) and b) number of species (s.e.) in each of the main faunal groups at Stringers Cove at the references at the start of fallowing and at the cage stations prior to stocking (TX), and at the start (T0) and end (T3) of the fallow period. The total number of species recovered is indicated in parenthesis beneath the site labels (X-axis) on Fig. b.

Table 5.4 Differences in abundance and Bray Curtis dissimilarity level of the ten species which most clearly distinguish between pre- (T0) and post-fallowing (T3) communities and between the pre-stocking (TX) and post-fallowing (T3) communities at Stringers Cove.

Species ID	*T0 (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissim. %*
Capitella capitata complex	27,470 (12,806)	5,525 (3,685)	1.20	29.13
Nebalia longicornis	14,902 (7,050)	1,791 (1,320)	1.58	25.50
Corbula gibba	353 (149.2)	222 (77.5)	1.30	3.40
Neanthes cricognatha	118 (53.4)	4 (1.6)	0.95	2.94
Nassarius nigellus	155 (87.3)	201 (42.0)	1.19	2.82
Birubius spp.	201 (79.5)	101 (23.1)	1.38	2.43
Theora fragilis	53 (45.0)	23 (6.2)	0.97	1.91
Echinocardium cordatum	43 (15.4)	71 (21.6)	1.32	1.67
Platyhelminthes sp.1	30 (15.7)	0	1.06	1.44
Mysella donaciformis	20 (10.5)	11 (3.7)	1.28	1.33

Species ID	*TX (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissim.%*
Capitella capitata complex	2 (1.1)	5,525 (3,685)	0.75	14.97
Nebalia longicornis	2 (1.0)	1,791 (1,320)	0.66	7.55
Corbula gibba	515 (232.8)	222 (77.5)	1.07	7.04
Echinocardium cordatum	112 (47.9)	71 (21.6)	1.80	3.48
Nassarius nigellus	151 (56.0)	201 (42.0)	1.74	3.29
Mediomastus australiensis	89 (50.9)	2 (1.3)	1.13	2.94
Birubius spp.	40 (16.0)	101 (23.1)	1.29	2.82
Nemertea sp.1	36 (23.0)	9 (4.4)	1.01	1.94
Aricidea sp.	66 (42.9)	0	0.68	1.94
Nemertea sp.2	9 (7.3)	36 (32.4)	0.77	1.92

Species ID	*Reference (No/m²)*	*T3 (No/m²)*	*Diss/SD ratio*	*Dissim.%*
Capitella capitata complex	19 (5.1)	5,525 (3,685)	0.75	10.38
Nebalia longicornis	0	1,791 (1,320)	0.66	5.49
Mediomastus australiensis	237 (51.8)	2 (1.3)	4.40	5.02
Nucula pusilla	180 (23.9)	2 (1.6)	3.28	4.52
Amphiura elandiformis	138 (10.1)	1 (0.6)	4.43	4.06
Lysilla jennacubinae	79 (14.5)	0	4.35	3.08
Echinocardium cordatum	0	71 (21.6)	1.52	2.90
Thyasira adelaideana	63 (8.2)	0	3.98	2.80
Birubius spp.	11 (5.5)	101 (23.1)	1.63	2.60
Corbula gibba	33 (7.9)	222 (77.5)	1.28	2.43

The post-fallowing community (T3) at Stringers Cove was very different to the references (ANOSIM Rho = 0.759, p = 0.01), but was closer to the pre-stocking community (ANOSIM Rho = 0.183, p = 0.04). The main differences between the post-fallow community and both the reference and the pre-stocking communities were the marked reductions in abundance of *Capitella capitata* complex and *Nebalia longicornis* (Table 5.4). In the pre-fallowing community these two species accounted for 59% of

the overall community similarity, but this was reduced to only 13% in the post-fallow community and neither species registered in the top ten species in the reference communities (Table 5.2).

Table 5.5 Proportional contribution of key ecological functions and reproductive strategies for the ten species which contributed most to the overall group similarity at the background (Reference) and post-fallowing communities (T3) communities

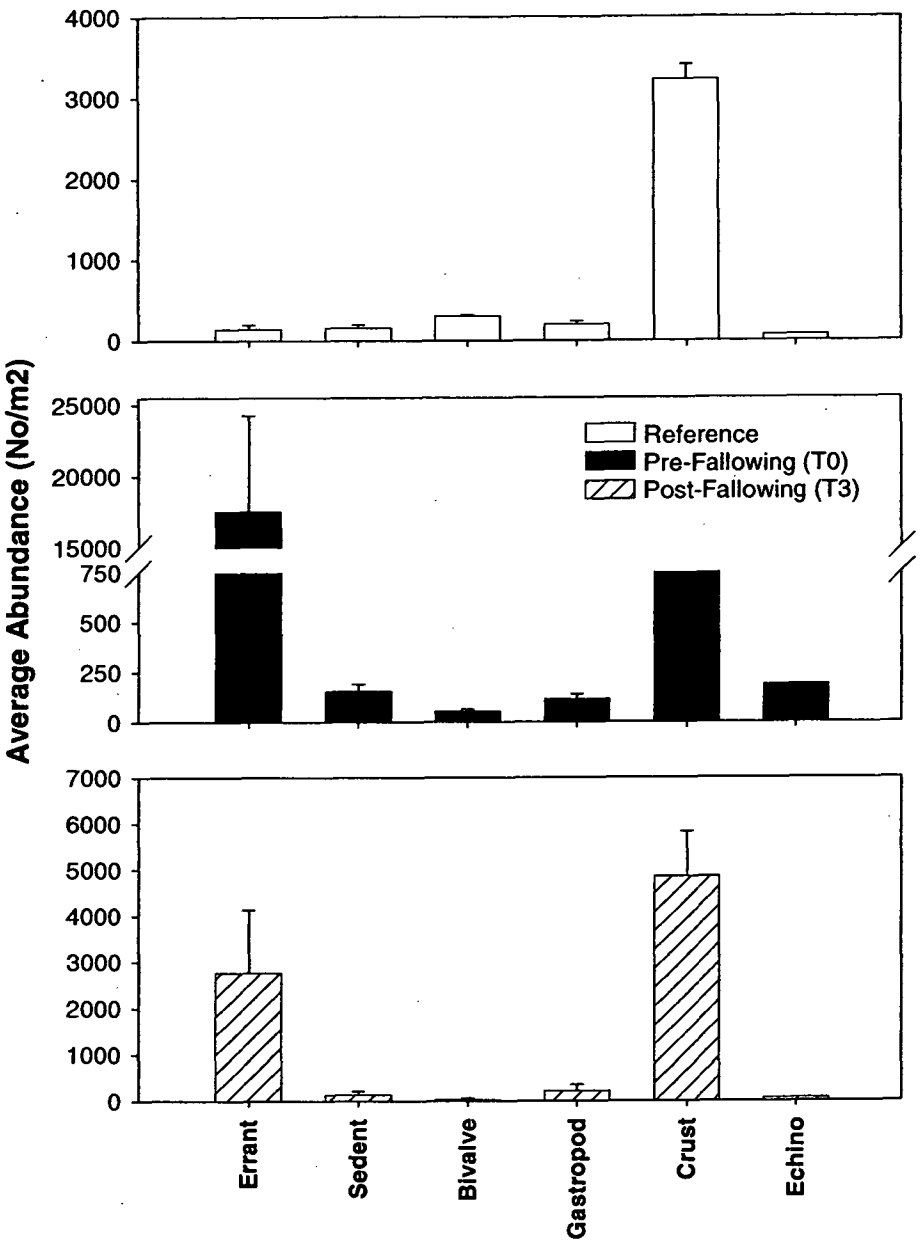
		Creeses Mistake		*Stringers Cove*	
Function		**Ref**	**T3**	**Ref**	**T3**
Size	*Small*	58	97	7	74
	Medium	34	1	73	26
	Large	8	1	20	0
Feeding Strategy	*Suspension Feeding*	22	0	6	0
	Deposit Feeding	59	95	74	100
	Carnivore	5	0	0	0
	Variable Feeding	14	5	20	0
Sediment Role	*Epibenthic*	0	1	0	2
	Stabilising	41	2	16	0
	Destabilising	59	97	74	98
	Unknown	0	0	0	0
Reproductive Strategy	*Staged*	39	5	39	4
	Opportunistic	20	89	20	92
	Unknown	41	6	41	4
Larval Development	*Pelagic*	11	1	11	27
	Benthic	73	8	73	3
	Benthic/ Pelagic	0	86	0	70
	Unknown	16	5	16	0

Both farm locations had species present with a range of functional and reproductive strategies, but, as with the species composition, there were some clear differences between the farms. The dominant species at the references at Stringers Cove had larger body size than at Creeses Mistake (Table 5.5). Although deposit feeding was the main feeding strategy at both farm sites, there were proportionally more suspension feeders in the reference fauna at Creeses Mistake (Table 5.5). The greatest change in the functional ecology of the dominant species occurred at the cage stations after farming (Fig. 5.6). Errant polychaetes were the group which most closely reflected the farming effects (Fig. 5.6). Post-farming these relatively small bodied mobile polychaetes, with an opportunistic reproductive strategy (i.e. *Capitella capitata* complex), dominated at both farm locations (Figs 5.6, 5.7 and 5.8). At Stringers Cove there was also an increase in crustaceans, primarily due to increases in abundance of the leptostracan *Nebalia longicornis* (Table 5.2). Over the fallow period, these small

opportunistic polychaetes markedly decreased in abundance at both farms, and at Stringers Cove the numbers of *Nebalia longicornis* also declined (Table 5.2). There was no significant difference in the bivalve to mollusc ratio over time at Creeses Mistake ($F = 1.40$, df 2, 21 $p = 0.269$), but at Stringers Cove the ratio was significantly lower at the end of fallowing than at the references ($F = 6.18$, df 2, 29 $p = 0.006$) (Fig. 5.6).

At Creeses Mistake there was very little difference in the community function at the reference sites over time. The greatest changes in functional and ecological response occurred at the cage stations over the fallow period. Principal components analysis shows that 95% of the variability in the sediment role and the feeding and reproductive strategies was associated with PC1, which reflects the separation of the farm stations over the fallow period (Fig. 5.7). At the start of fallowing (T0) the community at the cage stations was strongly dominated by deposit feeding sediment destabilisers/ bioturbators, and this remained the case until the end of the fallow period (Table 5.1 Fig. 5.7). In the post-fallowing communities the overall reproductive strategy did not change greatly, with opportunistic life strategies still proportionately more evident than at the references. At the end of the three month fallow period there was an increase in the abundance of species where the ecology is less well known (i.e. where the sediment interaction (US) and reproductive strategy (UR) was undefined), but the proportion of opportunistic sediment bioturbators/ destabilisers remained high.

a)


Average Abundance (No/m2)
4000
3000
2000
1000
0
25000
20000
15000
750
500
250
0
7000
6000
5000
4000
3000
2000
1000
0
Reference
Pre-Fallowing (T0)
Post-Fallowing (T3)
Errant
Sedent
Bivalve
Gastropod
Crust
Echino

b)

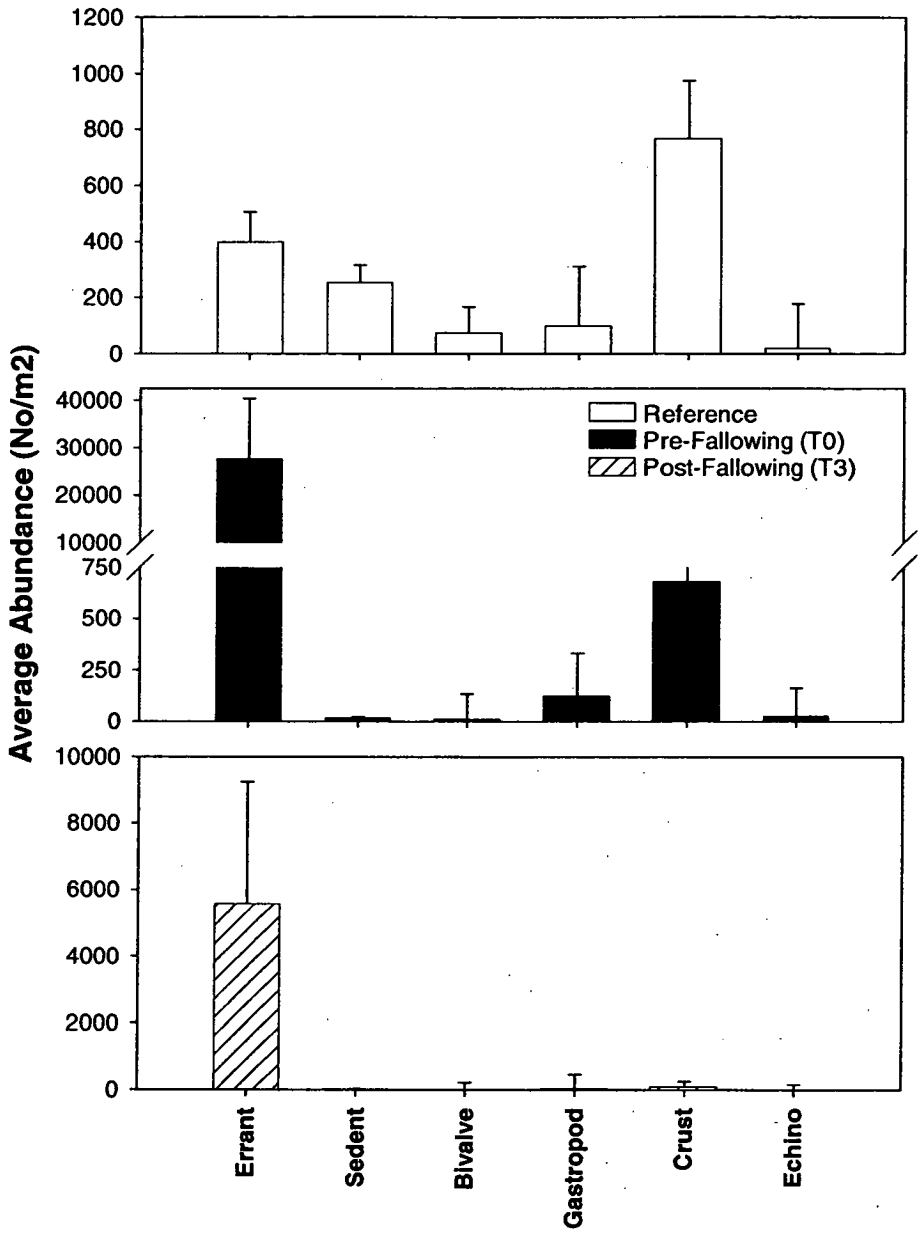


Fig. 5.6 Average abundance of faunal groups represented by ten dominant species at a) Creeses Mistake and b) Stringers Cove.

At Stringers Cove once again the main functional differences were associated with the cage stations, PC1 accounting for more than 96% of the overall variability in sediment role, feeding and reproductive strategies (Fig. 5.8). There was a clear gradient of change in function between the reference, post- and pre-fallowing communities (Fig.

5.8). The greatest functional change was between the reference and the pre-fallowing stations where there was a marked increase in the proportion of deposit feeders and sediment destabilisers in the pre-fallowing communities. There was also a change in reproductive strategy, to a community dominated by opportunistic species (Fig. 5.8). After fallowing (T3), there was some re-establishment of the fauna present at the references, but the suspension feeders and sediment stabilizers had not returned and the fauna still contained a large proportion of opportunistic species (Table 5.2, Fig. 5.8). The main functional differences between the pre- and post-fallowing communities were due to changes in reproductive strategy, the dominant species in the post-fallowing community tended towards benthic larval reproduction rather than pelagic larvae, which was the case at the reference communities (Fig. 5.8).

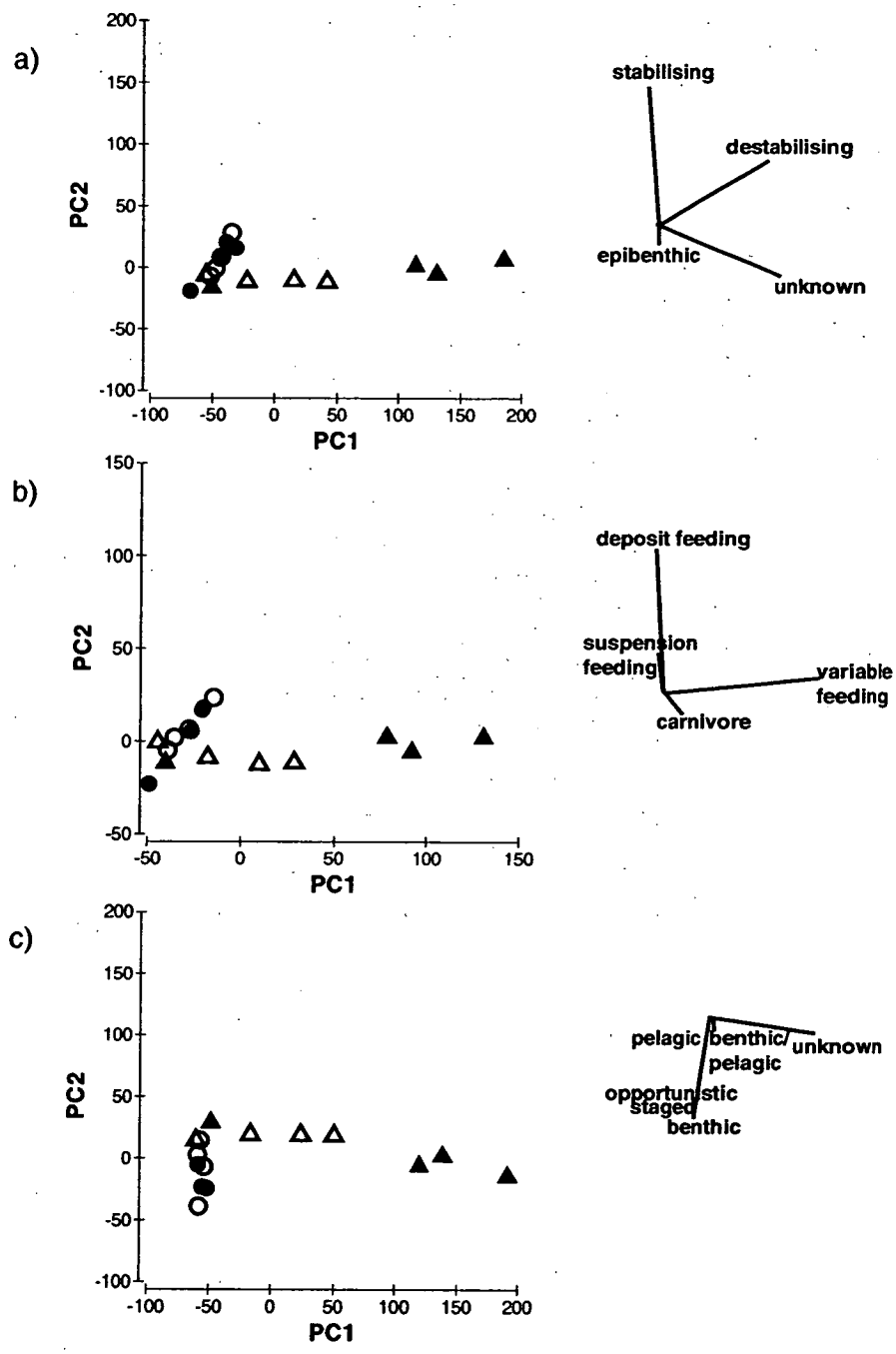


Fig. 5.7 Principal components analysis with biplot showing a) sediment role b) feeding strategy and c) reproductive strategy most strongly associated with site separations of reference ● (T0) and ○ (T3) and pre- ▲ (T0) and post △ (T3) fallowing stations at Creeses Mistake based on information for top ten species in each group. In all cases principal components 1 and 2 accounted for >99% of total variance.

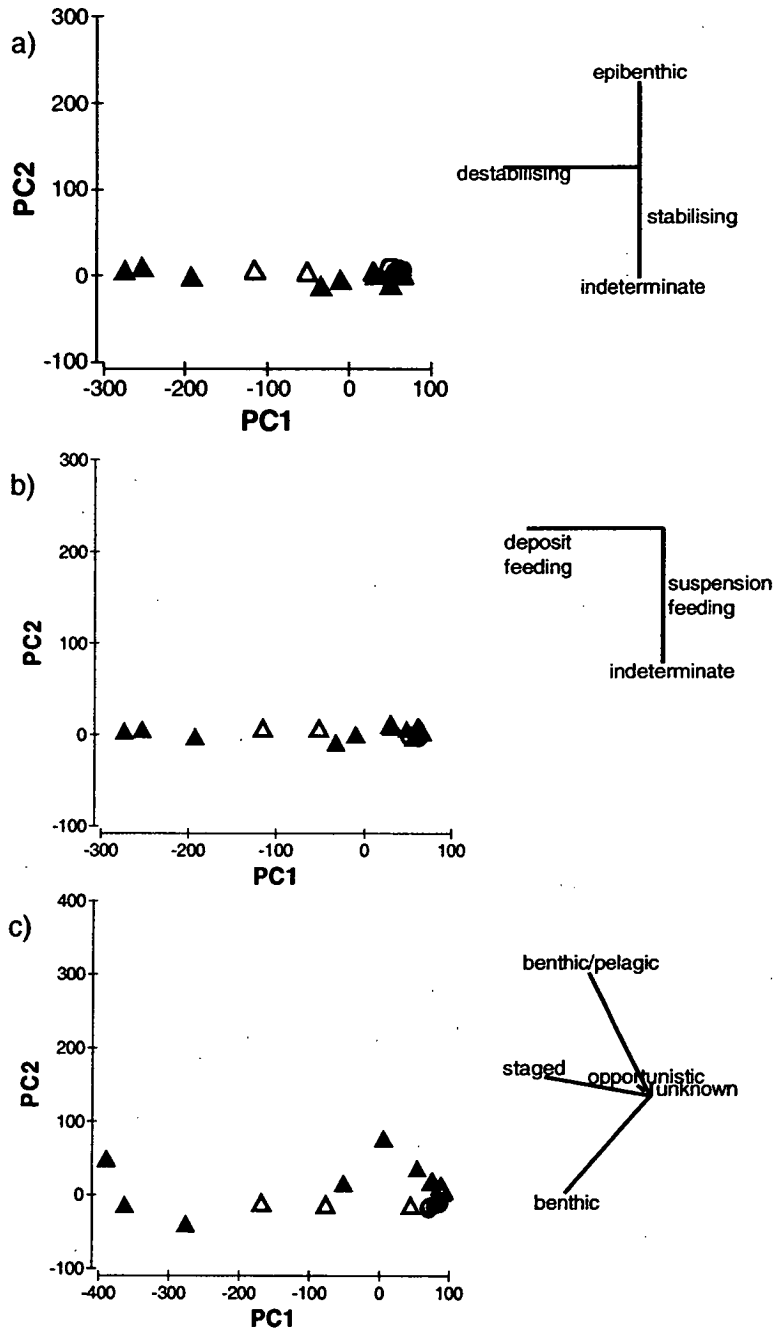
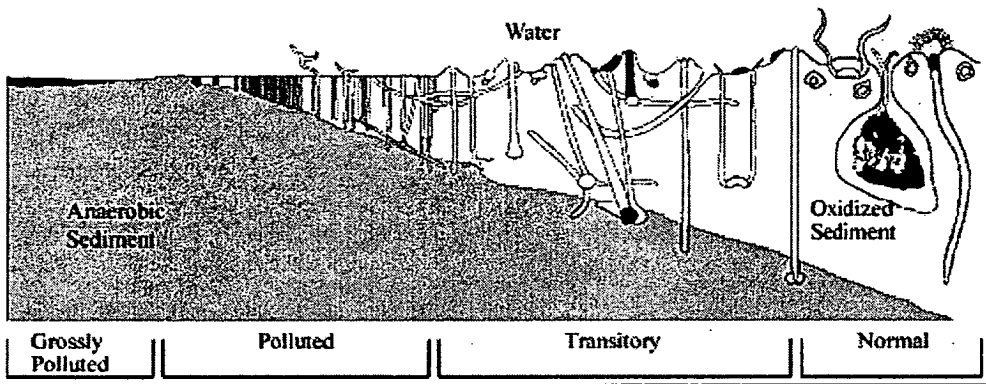


Fig. 5.8 Principal components analysis with biplot showing a) sediment role b) feeding strategy and c) reproductive strategy most strongly associated with site separations of reference ● (T0) and ○ (T3) and pre- ▲ (T0) and post △ (T3) fallowing stations at Stringers Cove based on information for top ten species in each group. In all cases principal components 1 and 2 account for >99% of total variance.

5.4 Discussion

There were marked differences in the recovery response at each of the two farm locations. There were changes in the infaunal community structure over the fallow period, but three months was not sufficient to restore the natural community structure at either location. At both locations the pattern of recovery was broadly consistent with the classical organic enrichment response model described by Pearson and Rosenberg (1978), albeit with locally specific indicators of the various successional stages (Fig. 5.9). The impacted communities were very similar in both systems, characterised by pollutant-tolerant opportunists, in this instance *Capitella capitata*, a small deposit-feeding polychaete indicative of organic enrichment (Grassle and Grassle, 1974, Pearson and Rosenberg, 1978). At both locations there was a progressional change in the community structure over the recovery period. However, there were major differences in the recovery response primarily as a result of the natural background conditions and resultant differences in the background communities at each of the farm locations.

In this study the structure of the background communities reflected the individual characteristics of each environment. Creeses Mistake is fully marine and relatively exposed with predominantly fine sand sediments whereas the Stringers Cove site is more sheltered with silt-clay sediments (Macleod *et al.*, 2002). In general, suspension feeders tend to be most abundant in high energy environments whilst deposit-feeders are more abundant in depositional areas with fine-grained muddy sediments (Snelgrove, 1999). Consequently, the ecology of the unimpacted communities was specifically and functionally quite different at each farm location; bioturbating and deposit feeding annelids were the predominant faunal group at Stringers Cove whilst Creeses Mistake had a greater proportion of suspension feeders and sediment stabilisers.



Dominant Genera in Pearson & Rosenberg model (1978)				
No Fauna	*Capitella*	*Labidoplax*	*Chaetozone*	*Nucula*
	Scolelepis	*Corbula*	*Anatides*	*Amphiura*
		Goniada	*Pectinaria*	*Terebellides*
		Thyasira	*Myriochele*	*Rhodine*
		Pholoe	*Ophiodromus*	*Echinocardium*
				Nephrops

Dominant Taxa in Current Study			
Creeses Mistake			
N/A	*Capitella capitata*	*Spionidae sp.2*	*Apseudes* sp.2
		Neanthes cricognatha	*Ampelisca* sp.
Stringers Cove			
N/A	*Capitella capitata*	*Nassarius nigellus*	*Amphiura elandiformis*
		Theora fragilis	*Mediomastus australiensis*
		Corbula gibba	*(Lysilla jennacubinae)*
		(Echinocardium cordatum)	*(Thyasira adelaideana)*

Fig. 5.9 Adaptation of Pearson and Rosenberg succession model (1978) showing local indicator taxa.

Although the community composition at Stringers Cove altered markedly after farming, the change in the ecosystem function was not great; the unimpacted and impacted communities were functionally quite similar, therefore functional recovery occurred relatively quickly. The functional significance of changes in the community composition depends on the species in question; some species may be lost without substantially altering the system function, whilst the loss of others may have serious consequences (Snelgrove, 1999). The dominant species at the reference stations at Stringers Cove were the brittle star *Amphiura elandiformis* and the polychaete *Mediomastus australiensis,* both of which like *C. capitata* are active bioturbators (Rosenberg *et al.*, 1997). *Amphiura* species, *A. eladiformis* in particular, are relatively intolerant of organic enrichment and low oxygen conditions (Pearson and Rosenberg,

1978, Crawford *et al*, 2002), but *Mediomastus* species are often found where organic content has been slightly enhanced. (Levin, 2000). Therefore, it appeared that the natural fauna at Stringers Cove may be better adapted to unconsolidated sediments and increased organic content than the fauna at Creeses Mistake.

The natural community at Creeses Mistake was strongly characterised by two suspension feeding crustacean species, the tanaid *Apseudes* sp.2 and the ampeliscid amphipod *Ampelisca* sp. When environmental conditions deteriorate crustaceans are often amongst the first members of the infauna to be affected (Nilsson and Rosenberg, 1994). The high levels of organic enrichment resulting from farming would overwhelm and eliminate this natural suspension-feeding community. Both these species are tube builders and as such recolonisation of these species would be constrained by the physical actions of a large deposit-feeding population which would both undermine sediment stability and clog filtering structures through sediment resuspension (Rhoads and Young, 1970). Accordingly, the natural community at Creeses Mistake would have much greater difficulty re-establishing than that of Stringers Cove and would be unlikely to exhibit functional recovery until the successional process was well underway.

In addition, for the community structure to fully recover the sediment conditions must be able to support the natural fauna. Recovery will be slower in areas where the natural fauna is unable to recolonise, either because the conditions are unsuitable for larval settlement or species immigration or because the reproductive cycles of the local species do not correspond to the recovery timeframe. Early colonisers play a critical role in ameliorating sediment biogeochemical conditions for subsequent species (Rhoads, 1974). Habitat condition plays an important role in determining settlement success and organic content is a specific sediment cue for many species (Butman *et al.*, 1988). The background organic carbon levels at Stringers Cove were relatively high (ca. 3%) (Macleod *et al.*, 2002) and consequently the natural fauna at Stringers Cove was pre-adapted to sediments with high levels of organic material. Although many of these species may have been overwhelmed by the additional organic loadings associated with farming activities, once farming ceased the environmental conditions would not be as inhospitable to these species and they would recolonise relatively

easily. The reproductive strategies of the native fauna at Stringers Cove were also well suited to rapid recolonisation, with a large proportion of the dominant species able to produce multiple generations of benthic larvae which could migrate directly to the recovering sediments from adjacent areas.

The background organic carbon levels at Creeses Mistake were very low (0.2%) reflecting the sandy nature of this site (Macleod *et al.*, 2002). After farming, the sedimentary carbon loading at Creeses Mistake had increased fivefold (Macleod *et al.*, 2002). The results indicate that after farming the natural suspension-feeding community was eliminated and that the feeding ecology had shifted to a community dominated by deposit feeders. Although species utilising different trophic modes can co-occur in large numbers, and distributions of suspension and deposit-feeders are not mutually exclusive (Snelgrove and Butman, 1994), bioturbation and sediment resuspension associated with large numbers of sediment deposit-feeders will often inhibit suspension feeding communities (Rhoads and Young, 1970, Brenchley, 1981). This may explain the faunal changes observed at Creeses Mistake and the elimination of the key stabilising species in the background community. For the natural community to re-establish at Creeses Mistake there would need to be a significant reduction in both the accumulated organic material and the abundance of bioturbating deposit-feeders. Several of the important species in the background communities at Creeses Mistake only reproduced at specific times of the year and this would also slow the recovery response by limiting the supply of larvae available for recolonisation.

There was greater diversity in the background fauna at Creeses Mistake, with almost 70% more species overall recovered at Creeses Mistake than at Stringers Cove. Areas with diverse communities tend to have a wider range of ecological functions, including species' mobility and reproductive strategies, and such communities will take longer to recover than those where diversity is low and the communities are simple (Thrush and Whitlach, 2001). Consequently, impacts will be more significant in areas, such as Creeses Mistake, with inherently high diversity (Thrush and Whitlach, 2001).

These findings have some interesting implications for locating and managing cage aquaculture operations specifically, but also for other more general organic

enrichment sources. The findings suggest that the premise that it is better for the environment to locate farms in more exposed locations to reduce the impact of organic enrichment by spreading the effects may be flawed. This study shows that, under similar farming impacts (Chapter 2), there was a greater change in the benthic infaunal community and ecosystem function at the more exposed location than at the more sheltered location and that the recovery response was slower. In addition, the overall area affected by organic deposition will be greater at exposed locations as compared with more sheltered sites because the current flow and or tidal influences are greater, thus increasing the field of dispersal. The fauna at more sheltered locations where organic-rich sediments accumulate may actually have a natural resilience to organic loading, being ecologically and functionally pre-adapted to cope with an increased level of organic enrichment. This suggests that, so long as the carrying capacity of system is not exceeded, sheltered locations may in fact be better suited to caged fish-farming. Finally, the differences in the recovery time (resilience) with location further reinforce the contention that managing recovery should take into account features of the receiving environment such as sediment type, organic matter content and ecological function (feeding strategies) of the resident infauna.

Chapter 6:

General Discussion

Understanding the recovery process is essential if the environmental effects of organic enrichment are to be managed effectively. Of the many factors that may influence the recovery response eight were considered in this thesis (Fig. 6.1), with the recognition that these are all strongly inter-connected and it is impossible to isolate the effect of any single factor. Chapters 2 and 3 evaluate the long-term recovery process whilst Chapters 4 and 5 examine recovery over the shorter-term (Fig 6.1 - Factor 3). The influence of the natural fauna (Fig 6.1 - Factor 1) on the recovery process has been specifically addressed in Chapters 3 and 5 for long- and short-term recovery respectively. Introduced species (Fig 6.1 - Factor 2) represented a significant component of the fauna, particularly in the early/transitional recovery phase, and the importance of these species in long and short-term recovery is discussed in Chapters 3 and 5. The site history (Fig 6.1 - Factor 6), nature (Fig 6.1 - Factor 5) and timing (Fig 6.1 - Factor 4) of any given impact have a marked influence on both the short- and long-term recovery response and consequently these factors are discussed throughout the thesis. Exposure (Fig 6.1 - Factor 7) and sediment conditions (Fig 6.1 - Factor 8) are strongly linked, both to each other and to the site history and as a result these factors are also discussed throughout the thesis. However, differences in exposure and sediment condition were of particular interest in the short-term recovery studies (Chapters 4 and 5), where a specific comparison was made between locations with very different environmental conditions. Clearly this study has markedly advanced our understanding of the relationships between these factors and the processes underpinning recovery from organic enrichment in southern temperate conditions.

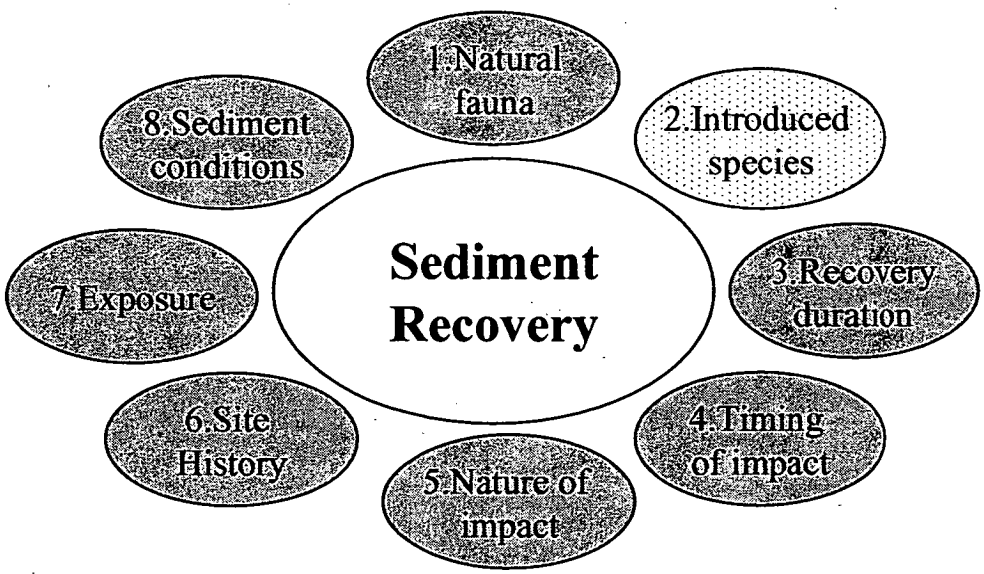


Fig. 6.1 Model of main factors influencing recovery process showing factors that have been addressed in the current study.

6.1 Estimation of Recovery

In undertaking this research a distinction was made between long-term recovery, i.e. that generally associated with one-off enrichment events, and short-term recovery, more commonly associated with recurrent impacts. Long-term recovery considers the recovery process associated with the passive re-establishment of the natural communities over a period of several years. In contrast short-term recovery examines the changes in the macrofaunal communities between recurrent impacts, where the timeframe for recovery is short. Although it is important to know the length of time required for total recovery, in many cases establishing the length of time for lower levels of recovery is probably just as important. For example, in the aquaculture industry, particularly in Tasmania, it is a common operational practice to leave areas of the sea bed clear from farming operations for relatively short periods of time to promote sediment remediation.

The present results suggest that evaluation of the functional status of the system may be a more useful approach for managing recurrent organic enrichment of the kind associated with aquaculture operations in Tasmania. Functional status is more useful in

regional comparisons and can more readily be used to define regional standards. In the present study the functional mechanisms examined included the main sediment bioturbation mechanism, and the primary feeding and reproductive strategies of the benthic macrofauna. Defining the system in this manner provides a much better understanding of the way in which the system is operating. However, the value of any measurement of functional recovery will in large part depend on how well the fauna is known. It should be no more difficult to undertake than full community assessment, but is arguably more meaningful to environmental managers and non-ecologists than just a list of species or taxa with no particular significance.

Although the overall recovery response in southern temperate climes appears to follow the classical successional models defined by Pearson and Rosenberg (1978) and Rhoads *et al.* (1978) for northern temperate regions, there were regional differences between sites. Critically there were differences in communities in the unimpacted site between the study farms due to differences in the prevailing environmental conditions. In some areas the background organic content was naturally enhanced and the fauna reflected these conditions with the reference community exhibiting a faunal composition more akin to the transitory communities in the northern hemisphere. The presence of infaunal assemblages with differing successional endpoints between regions was recognized by Zajac and Whitlach (1982), but differing endpoints may also occur over smaller spatial scales, even within systems (Rhoads and Germano, 1986). Zajac (2001) describes a series of community types (I, II and II) which are similar ecologically to the polluted, transitory and normal communities as described by Pearson and Rosenberg (1978) (Figure 5.9), but which represent potential successional endpoints or climax communities that can persist over time as a recognisable community type. Recovery from organic enrichment where the successional endpoint is a Type II community may proceed more quickly than where the endpoint is a Type III community. Recolonisation capacity and the ensuing successional pathways will be shaped by factors such as seasonality and life history (Zajac, 2001). In a Type II community there are more likely to be species with life history strategies suitable for rapid recolonisation through immigration, (tolerant species) and colonization (more flexible reproductive strategies). This appeared to be the case at Stringers Cove

(Chapter 5) where a significant component of the background fauna was able to produce both multiple generations and benthic larvae capable of direct migration to the recovering sediments. Consequently the timing of the recovery phase can be critical in areas where the natural species life histories are strongly influenced by seasonal cycles whereas this would have less affect where direct migration is the main recolonisation process.

In marine finfish farming it is accepted that exposed locations represent better environments for farming as the organic waste materials are dispersed over a wide area, with the result that there is a lower impact in the vicinity of the cages. However, the results from the present study suggest that this premise is flawed. Although it is true that the organic waste material will be more dispersed in areas with higher exposure and current flow, it also means that a wider area will be affected at a low level. The findings of this study showed that the faunal community associated with the more exposed location was less resilient to organic impacts; the fauna was less able to tolerate and assimilate organic content and was less well adapted for recolonisation of organically enriched sediments than the fauna at more sheltered locations (Chapter 4 and 5). Fauna at the sheltered location were better adapted to assimilate the organic inputs from fish farming and therefore more resilient to the impacts.

Recolonisation and recovery of the infauna commenced as soon as organic inputs stopped (Chapter 3 and 5). Amongst these initial colonizers were the typical opportunists but also species with tolerance to high levels of organics in the sediments as well as epibenthic scavengers able to exploit this abundant food resource whilst avoiding the deteriorated sediment conditions. The main successional changes in the community function were also outlined in the classical response models (Pearson and Rosenberg, 1978, Rhoads *et al.*, 1978). This suggests that the function of the system was largely restored after 12 months (Chapter 5). In relation to determining the sustainability of aquaculture operations functional recovery appears to be a better benchmark against which to judge recovery than full community assessment (Chapter 3 and 5). Once the system is functionally restored it has the capacity to recover to a stable state given sufficient time and therefore recovery could be considered sustainable.

Where recovery was undertaken over the longer term (i.e. several years) the ecological response followed that outlined in the successional models of Pearson and Rosenberg (1978) and Rhoads *et al.* (1978) (Chapter 3). Three distinct ecological phases were identified; an unimpacted phase equivalent to the stage I (Rhoads *et al.*, 1978) or unpolluted conditions (Pearson and Rosenberg, 1978), an impacted phase equivalent to stage III (Rhoads *et al.*, 1978) or polluted conditions (Pearson and Rosenberg, 1978) and an intermediate phase equivalent to the stage II (Rhoads *et al.*,1978) or transitory community (Pearson and Rosenberg, 1978) (Chapter 1 - Fig. 1). In contrast sites exposed to repeated impacts and only allowed short recovery periods only achieved stage II or transitory communities. The most polluted conditions were characterized by *Capitella capitata* which is as strong an indicator in the southern hemisphere as it is in the northern. Several other genera or taxonomic groups common to unimpacted and transitory conditions in northern temperate systems were also represented in the southern temperate communities, in particular *Amphiura* and *Nucula*. However, there were also some distinctly local indicators i.e., *Malacoceros tripartitus* in impacted conditions, the local dog whelk *Nassarius* in the transitory conditions, often where organic content was elevated and *Lysilla jennacubinae*, the tanaid crustacean *Apseudes* sp and the amphipod *Ampelisca* were all variously associated with unimpacted conditions at different locations. Species of the genus *Thyasira* had been described as a transitory species in the northern communities but in these local studies *Thyasira adelaideana* was associated with unimpacted conditions at one location. This may reflect the differences in the background conditions at the sites. Other local studies (Crawford *et al.*, 2002, Edgar *et al.*, 2005) have also found *Nassarius* in areas of increased organic enrichment, whilst a recent study of aquaculture in Israel identified members of the genus *Nassarius* associated with cage operations (Angel *et al.*, 1998).

Several introduced species were prevalent in the infaunal communities, particularly the transitory communities. Introduced species by nature are often extremely tolerant of disturbed conditions. It has been suggested that unlike local opportunists these species do not necessarily follow predicted successional patterns. Once established they have the capacity to significantly alter local ecology, gain a foothold and expand their distribution (Ruiz *et al.*, 1999). However, in the current

study the responses of these species were consistent with expected successional models. Rather than dominating recovering communities, abundances of introduced species tended to decline as sediment conditions improved. The monitoring programs associated with cage aquaculture or other organic enrichment sources represent an important avenue to further examine the ecological interactions of these species.

The relationship between organic loading and sediment recovery is not a simple one. The correlation between the impact level indicated by the sediment geochemical response to organic enrichment and that indicated by the ecological response was very different in the southern systems as compared with that proposed in the northern temperate models (SEPA, 1998, Wildish *et al.*, 1999, Levings *et al.*, 2002). Biological changes were apparent at much lower levels of impact in the southern systems (Chapter 2). The initial impact is one of the major factors affecting recovery level and the length of time required for recovery is positively related to the size of the disturbance (Hall *et al.*, 1994). However, it appears that impact level is very strongly influenced by the background conditions (Chapter 4).

Chemical remediation of the sediments was very rapid, with a marked improvement in the first 2-3 months, which is consistent with the findings from many other studies (eg. Weston, 1990, Brooks *et al.*, 2003). However, sediment biogeochemistry was generally not found to be a useful approach for monitoring. Sulphide levels were substantially lower than might be expected whilst organic matter contents were higher (Chapter 2). Organic content was very poorly correlated with impact and recovery levels. This was largely a result of the way in which these approaches measure organic content. The long-term study location and one of the short-term locations (Stringers Cove) had inherently high organic matter loadings. However, much of this material was not associated with the organic enrichment resulting from farming. It was derived from terrestrial sources and was refractory in nature, and therefore was unavailable to the fauna (Macleod *et al.*, 2004b). Since determination of organic content by loss on ignition and total carbon was unable to distinguish this refractory component (Chapter 2), it was not found to be a useful predictor of either total organic inputs or the successional stage of the fauna. Sulphide levels were more consistent between locations, however, overall sediment sulphide

content was relatively low and recovered very quickly, dropping to undetectable levels within the first few months. Even the highest sulphide levels in present study fell within northern guidelines (SEPA, 1998, Levings *et al.*, 2002). Overall, the sediment geochemistry did not response in a manner comparable to that predicted using northern temperate models (Pearson and Rosenberg, 1978, Wildish *et al.*, 1999). In general the sediment geochemistry recovered rapidly and did not reflect the recovery indicated by the ecological succession. Consequently it is concluded that although measurement of redox and sulphide may be useful for assessing impact is not appropriate for evaluation of recovery.

Both the biological and the geochemical results have shown that although there are strong similarities in the manner in which the sediments recover between the northern and southern temperate regions, there are still distinct geographical differences. Although functional succession was similar there were many taxonomic distinctions in the community succession. The geochemical results particularly highlight the geographical differences between the northern and southern hemispheres. They clearly show that although trends may be similar, absolute levels can't be extrapolated over large spatial scales and that it is necessary to establish local baseline data to manage local conditions.

6.2 Management Implications

Although there were differences in the recovery rate and response as a result of background environmental conditions, it was determined that there were components of the recovery response that could be predicted once the baseline conditions were understood (Chapter 4). Under the production levels employed in this study a fallow period of three months was not sufficient for complete recovery at either location, but the findings did show that recovery to pre-stocking levels was possible in 3 months. The recovery response was quite different at locations with different background environmental characteristics, with recovery beginning slowly before accelerating at one location whilst at the other recovery was initially rapid and then slowed. Consequently shortening the fallow period would have a greater effect at locations where recovery was initially slow. The results also suggest that there may be an

optimum level of impact above which the extent of impact decreases more rapidly and the level of recovery is reduced but below which the level and extent of recovery are similar. These findings show that understanding the circumstances at any given location is critical to determining an appropriate timeframe for recovery and that the optimal length of fallow period needs to be determined on a site by site, or even cage by cage basis. This represents a significant step forward in understanding and managing recovery.

Many different techniques have been used as surrogate measures of sediment condition (Hargrave *et al.*, 1997; Kingsford and Battershill, 1998). These include measurement of redox and sulphide, total organic matter and/or total organic carbon levels and more commonly these days, visual assessment of sediment characteristics and epibiota by diver or video (Angel *et al.*, 1998, Crawford *et al.*, 2002). Measurement of sediment chemistry showed that redox and sulphide were poor indicators of sediment recovery and that organic carbon levels were difficult to interpret given that many of the local sediments contained high levels of refractory organic material. Video assessment was an effective means of viewing recovery, but it also indicated a more rapid recovery than the benthic community.

6.3 Conclusions

This study has greatly increased our understanding of the recovery processes associated with organic enrichment in southern temperate regions. The ecological response of the system was shown to be a more useful measure of recovery than changes in production levels or geochemical conditions. Evaluation of the functional response of the system was determined to be the best approach for monitoring recovery where recurrent organic enrichment is occurring (i.e. aquaculture operations). However, such monitoring programmes must be carefully evaluated, or further studies undertaken, to ensure that progressive deterioration is not occurring.

The results indicate that, since recovery response differs depending on the background environmental conditions, establishment of baseline conditions and local benchmarks is essential in evaluation of impact and recovery, for establishment of a

regulatory framework and for ongoing environmental management. However, these baselines and the subsequent management protocols must be established at a spatial scale relevant to the community (ecological) changes. In the context of cage aquaculture operations this may mean on a lease by lease, if not cage by cage, basis.

6.4 Further Research

There were spatial differences in the recovery response associated with the specific nature of the background communities and sediments. This suggests that within areas, or even between areas, with similar sediment structures the recovery response may be similar. In a recent study of the impacts of fish farming over 20 locations in southern Tasmania it was suggested that responses could be characterized regionally according to sediment characteristics (Edgar *et al.*, 2005). Macleod and Helidoniotis (2004) also characterized 11 distinct infaunal community types which could be associated with changes in sediment condition from 80 study sites within the two largest estuaries in southern Tasmania. Previous studies have indicated that it is not a trivial exercise to extrapolate recovery from smaller to larger scales and that caution should be exercised in such assessments (Reise, 2001, Thrush and Whitlach, 2001. However, if it is possible to categorise sediments and their associated communities according to their likely response (resilience) to organic enrichment, this would be extremely useful information from a management perspective and as such should be investigated.

One area that was not investigated by this study is the likelihood of progressive deterioration resulting from repeated impacts. The findings showed that different recovery responses could be achieved with different recovery time periods and under different environmental conditions (Chapter 3 and 4). It is important that farm managers continue to monitor their individual leases/individual cages to ensure that progressive deterioration does not occur and further research is required to identify the risk factors that could potentially lead to sediment souring.

Several well known introduced species were present in the various stages of the recovering communities in this study i.e. *Corbula gibba*, *Maoricolpus roseus*, *Theora*

lubrica and *Euchone limnicola*. Introduced pests are internationally recognised as a serious threat to marine biodiversity, second only to habitat loss (Baltz, 1991). Exotic species have the capacity to significantly affect the native ecology. To be a successful introduced species they must be plastic in their habits and tolerances, and therefore able to exploit any environmental niche (Carlton *et al.*, 1990). However, it is not possible to predict how these species might specifically affect the successional response as there is currently very little information on the interactions of introduced species in the local environment.

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